Modelling adaptation strategies for Swedish forestry under climate and global change

Víctor Blanco González

A thesis presented for the degree of

Doctor of Philosophy

University of Edinburgh

2016

Own Work Declaration

I hereby declare that this thesis was composed by myself, that the work contained herein is my own except where explicitly stated otherwise in the text, and that this work has not been submitted for any other degree or professional qualification.

V. Blanco González

July 2016

Acknowledgements

There is a great number of friends, colleagues and acquaintances that deserve being acknowledged here for their help and support during this three and a half year journey. While I am unable to mention them all here, I hope that they all know how very grateful I am.

This project has largely been the result of a team effort between colleagues at the University of Edinburgh and the Mistra-SWECIA Programme. First and foremost, I feel obliged to my supervisors: Mark Rounsevell, for being immensely supportive and inspiring throughout, and for sharing his large expertise; and Calum Brown, for his great commitment, patience, enthusiasm, encouragement, and his numerous useful comments and valuable advice. I am also thankful to Sascha Holzhauer for sharing his expertise, and for his endless patience during the process of getting the model to work. Collegues from Mistra-Swecia brought in multi-disciplinarity, a great pool of knowledge, ideas and support, which helped to bring forward this project, but also helped me to grow professionally and as a scientist.

The University of Edinburgh excited and stimulated me from the moment I arrived, and it has contributed numerous oportunities for professional growth. For their financial support I express my gratitude to the Mistra-SWECIA Programme and the University of Edinburgh.

For their unconditional friendship and support I would like to thank my friends in Edinburgh and those that I left behind when coming here, but who have been 'there' nevertheless. Special thanks to Dahyann, who has helped me to overcome the setbacks, patiently beared my frustrations, and celebrated every triumph, big or small. Finally, to my family, and especially my parents, who give me their unconditional support in everything that I do. Thank you.

Table of Contents

List of Figures

List of Tables

List of Abbreviations

ABM	Agent-Based Model
CRAFTY	Competition for Resources between Agent Functional TYpes
ES	Ecosystem Services
FAO	Food and Agriculture Organization
FMFT	Forest manager functional types
IPCC	Intergovernmental Panel on Climate Change
LARA	Lightweight Architecture for boundedly Rational Agents
RCP	Representative Concentration Pathways
SDD	Supply-Demand Difference
SEPA	Swedish Environmental Protection Agency
SFA	Swedish Forest Agency
SES	Socio-Ecological System
SSP	Shared Socio-economic Pathways
UNECE	UN Economic Commission for Europe
MCPFE	Ministerial Conference on the Protection of Forests in Europe
FSC	Forest Stewardship Council
PEFC	Pan-European Forest Certification Council

Thesis Abstract

Adaptation is necessary to cope with, or take advantage of, the effects of climate change on socio-ecological systems. This is especially important in the forestry sector, which is sensitive to the ecological and economic impacts of climate change, and where the adaptive decisions of owners play out over long periods of time. These decisions are subject to experienced and expected impacts, and depend upon the temporal interactions of a range of individual and institutional actors. Knowledge of, and responses to, climate change are therefore very important if forestry is to cope with, or take advantage of, the effects of climate change over longer timescales.

It is important to understand the role of human behaviour and decision-making processes in the study of complex socio-ecological systems and modelling is a method that can support experiments to advance this understanding. This study is based on the development of CRAFTY-Sweden; an agent-based model that allows the exploration of Swedish land-use dynamics and adaptation to climate change through scenario analysis. In CRAFTY-Sweden, forest and farmland owners make land use and management decisions according to their objectives, management preferences and capabilities. As a result of their management and location characteristics they are able to provide ecosystem services. To explore future change, quantitative scenarios were used that considered both socio-economic development pathways and climatic change. Simulations were run under the different scenarios for the period 2010-2100, for the whole of Sweden. Furthermore, because institutions (i.e. organisations) also influence socio-ecological systems through their actions and interactions between them and with land owners and the environment, a conceptual model of institutional actions applied to socio-ecological systems was developed. The application of this conceptual model was explored through a model of institutions that can act, interact and adapt to environmental change in attempting to affect ecosystem service provision within a simple forestry governance system.

I found that forestry in the future will likely be unable to meet societal demands for forest services solely on the basis of autonomous adaptation. A northward expansion of agriculture and especially of forestry proved positive for both sectors to adapt to changing conditions, under several scenarios, given the substantial land availability and the improved environmental conditions for plant growth. Legacy effects of past land-use change can have a great impact on future land-use change and adaptation processes, especially in forestry. Also, greater competition for land may lead to shorter forest rotation times. Socio-economic change and land owner behavioural differences may have a larger impact on owner competitiveness, land-use change and ecosystem service provision than climate-driven changes in land productivity. Different owner objectives and behaviour resulted in different levels of ecosystem service provision. Also, particular forest types were differently suitable for adaptation depending on the sets of objectives under which they were managed. Owners implementing particular management strategies can be differently competitive under different future scenarios, and the suitability of such strategies for adaptation is not a static, inherent characteristic of a system. Instead, it evolves in response to changing contexts that include both the external global change drivers and the internal dynamics of agent interactions. Additionally, institutional conceptual models as presented here can support better understanding of the key institutional decision-making dynamics and their consequences, endogenously, flexibly across different socio-ecological systems. Finally, study limitations, future research and the policy relevance of findings are discussed.

Lay Summary

The uncertain effects of climatic change and changing demands for ecosystem services on the distribution of forests and their levels of service provision require assessments of future land-use change, ecosystem service provision, and how ecosystem service demands may be met. This is especially so in countries such as Sweden, which have large forest areas that are economically and culturally important, and which are likely to be affected by climatic change. Adaptation is necessary to cope with or take advantage of the effects of climate change, especially in the forestry sector, where the adaptive decisions of owners play out over long periods of time. Under such uncertain prospects there is an obvious need for in-depth studies of potential future land-use transitions in Sweden with a focus on forestry, to better understand possible changes in forest management and ecosystem service provision. The main questions answered with this research are: 1) What owner types and management strategies exist in the Swedish forestry sector and how are their management decisions made? 2) How might global change influence future land use change and ecosystem service provision in Sweden? 3) How can the forestry sector adapt to environmental change in meeting future demands for ecosystem services in Sweden? 4) How can institutions, their actions and interactions in the forestry sector be modelled? I explore ecosystem service provision, land-use change and adaptation to global change in the forestry sector using CRAFTY-Sweden, an agent-based model that represents large-scale land-use dynamics, based on the demand and supply of ecosystem services. Services are supplied by land owners on the basis of their objectives, management preferences and other behavioural traits, as they compete for land under changing environmental conditions. Future impacts and adaptation within the Swedish forestry sector were simulated for scenarios of socio-economic change and climatic change, between 2010 and 2100. Furthermore, because institutions (i.e. organisations) also influence socio-ecological systems through their actions and interactions between them and with land owners and the environment, I developed a conceptual model of institutional actions applied to socio-ecological systems. I discuss the results in terms of their implications for the modelling of socio-ecological systems and adaptation. I also examine the effects of different drivers on ecosystem service provision and on the suitability of different management strategies for adaptation, and suggest how future societal demands

for forest services can be met by means of individual and sectoral adaptation to global change.

Chapter 1

General Introduction

1 General introduction

Land-use activities have transformed a large proportion of the planet's land surface (Foley et al. 2005). Such land is today under intense pressure, subject to the demands of a growing human population and to changing patterns of consumption (Bouma et al. 1998; Godfray et al. 2010; Smith et al. 2010). These demands drive competition for the limited land resource between food producers, resource extractors, nature conservationists or urban developers amongst others. On top of this, the effects of climatic change on land systems are already being seen (e.g. Larson 2013; Parmesan and Yohe 2003; Soja et al. 2007), and are expected to become greater (IPCC 2014b).

Globally, forestry production (i.e. timber and non-timber forest products) is estimated to change only modestly with climate change, albeit regional and local changes are likely to be large (IPCC 2007; Soja et al. 2007). Forest productivity increase is generally expected to happen in the long term especially at high-latitude regions, although recent studies have also shown declining productivities in several areas of boreal forest, attributed principally to warming-induced drought (IPCC 2014b; Schroter et al. 2005). Such regional changes are expected to be particularly apparent in Sweden, where forests have great economic, environmental and cultural importance. There, significant projected future changes include an increase in the frequency of warm temperature (Fig. 1) and precipitation extremes, and a weakening of cold extremes (Rummukainen et al. 2012; Schroter et al. 2005). Forests are one of Sweden's most valuable natural resources, covering 69% of the country, and providing for one of Sweden's largest industries, the forestry sector (SLU 2015; Ulmanen et al. 2012). In 2010, the forest industry accounted for 10–12% of total employment and turnover in Swedish industry, and 11% by value of Sweden's exports. In several counties the forest industry accounted for 20% or more of industrial employment. The sector accounts for about 3% of Sweden's gross domestic product.

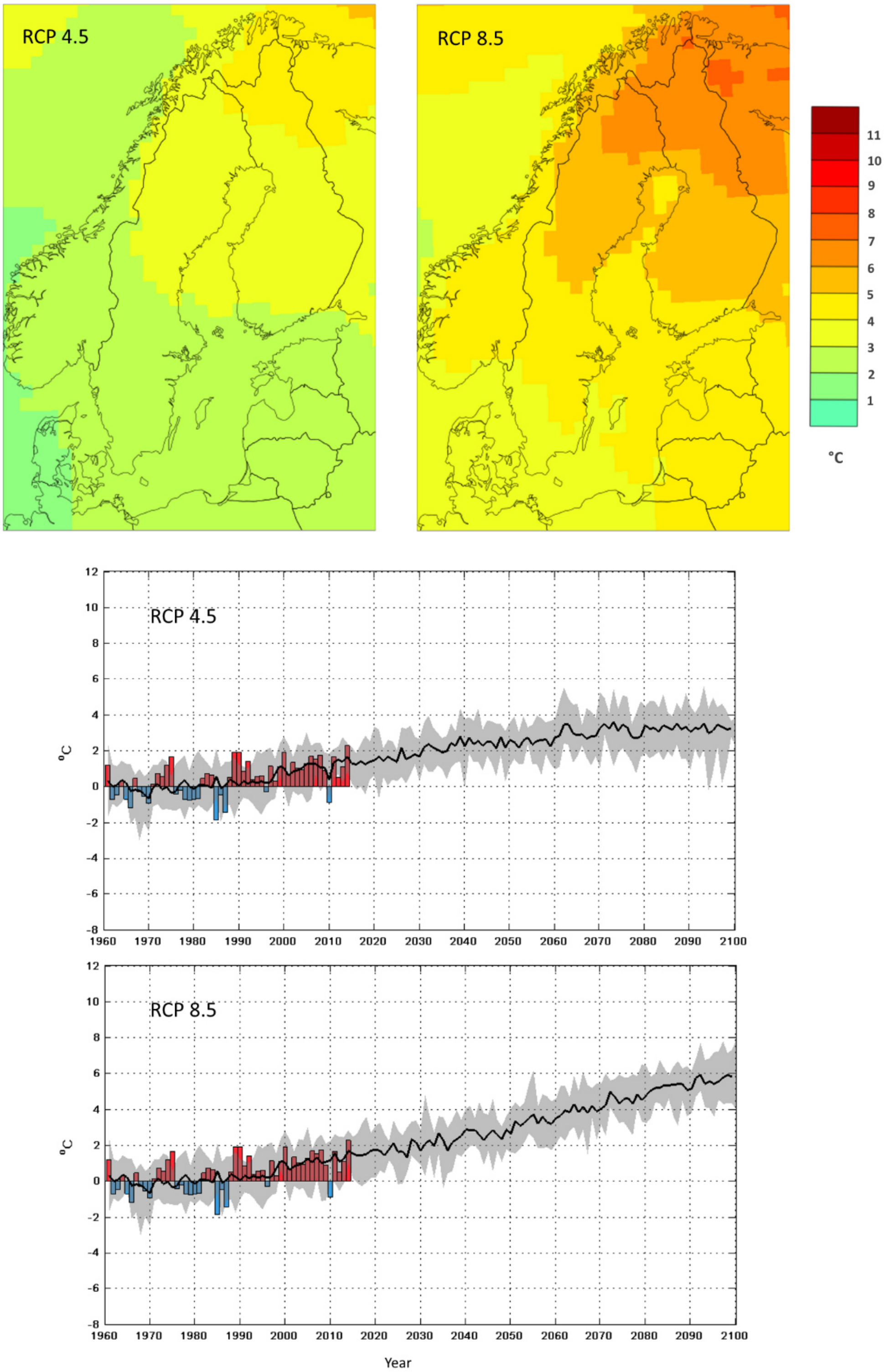


Fig. 1 Changes in annual mean temperature in Sweden under Radiative Concentration Pathway (RCP) 4.5 (moderate increase) and 8.5 (high increase) (adapted from SMHI (2016)).

Swedish forestry is expected to be significantly affected by climate change, in terms of increased forest growth and increased forest damage and biodiversity loss (IPCC 2007; IPCC 2014a; Schroter et al. 2005), and ecological responses and adaptation (Sedjo 2010). There are concerns that indirect effects of climate change can make forests more sensitive to strong winds, as long as management regimes remain unadapted, if trees grows taller, if they are stressed by other weather extremes or from not being acclimatized to the new climate, or if ground frost melts (Blennow et al. 2010). Bark beetle outbreaks and damage are often a secondary impact following the wind throw of trees (Lagergren et al. 2012). Additionally, demand is anticipated to substantially exceed the potential supply of woody biomass in Europe up to 2030, putting a very high pressure on Swedish forest resources and likely forcing difficult trade-offs between forestry policy goals (Jonsson et al. 2011). At the same time, land and resource use is expected to have an even greater effect on biodiversity than climate change (Swedish Biodiversity Centre 2008; as cited in Ulmanen et al. 2012). Furthermore, the fact that important forest owner decisions, such as those regarding what species to plant, play out over several decades makes the forestry sector particularly vulnerable to socio-economic and climatic change. In the midst of such uncertain changes, there is a need to explore how the forestry sector may be affected by socio-economic and environmental change (i.e. global change) in the future, and how the sector may be able to adapt in order to meet societal demands for ecosystem services (ES).

Sweden has established greenhouse gas emission reductions as part of its commitments made under the United Nations Framework Convention on Climate Change (UNFCCC). The Swedish milestone target for the environmental quality objective 'Reduced Climate Impact' states that emissions (from activities not included in the EU Emission Trading System) are to be reduced by 40%, or around 20 million tonnes carbon dioxide equivalents (Mt CO_2-eq.) between 1990 and 2020 (Swedish Environmental Protection Agency 2016). Removals and emissions from forestry and other land uses are, however, currently not included in the national target (Swedish Environment Agency web 2016). This allows Sweden flexibility to plan future land use that is independent of the country's mitigation targets.

It is expected that Sweden will achieve its 2020 target. Swedish emissions, excluding net CO_2 from land use, land-use change, and forestry (LULUCF), were reduced by 14.14 Mt CO_2-eq. between 1990 and 2014 (Swedish Environmental Protection Agency 2016). LULUCF contributed in 2014 a net uptake of 45.07 Mt CO_2-eq., making the country carbon neutral as of the same year. Forestry, and particularly standing forest biomass, makes the majority of the carbon uptake accounted for LULUCF. Net removals in this sector are heavily influenced by harvests and natural disturbances (e.g. storms). Nevertheless, the carbon emissions considered when setting reduction targets only included in-house emissions, while emissions associated with overseas production of goods and services imported and consumed in Sweden are not included. Sweden is a net importer of CO_2, importing 11 Mt CO_2-eq more than it exported in 2008 (Minx et al. 2008). If such emissions were to be considered in the future as part of net emission reduction targets, it is possible that LULUCF could be included. This being the case, it becomes essential to consider the expected substantial reduction in forest carbon sequestration that will take place before 2050, associated with large-scale harvests. Such oscillations can have a major impact on the uptake of carbon in Sweden, and the effects can last for decades, given the relatively slow growth rates of forests.

Sustainable forestry is one of the 16 environmental quality objectives to be achieved by 2020 (Swedish Environmental Protection Agency 2012). The Swedish Environmental Protection Agency states that:

“To preserve important forest environments, nature reserves and other forms of protection are needed, combined with voluntary set-aside of forest land by owners. Forest areas may also need to be restored or managed in ways that enhance their values. […] A broader challenge is to adapt forestry practices so that they conserve and develop the natural and cultural values of forests, while still remaining competitive.”

To achieve sustainable forestry, Swedish Environmental Protection Agency sets specific milestone targets for nature reserve and voluntary set-aside expansion (Swedish Environmental Protection Agency 2016): In relation to forests, first, an increase is necessary in formally protected forest land of approximately 150 000 hectares of high nature value below the montane forest zone. Additionally, voluntary set-aside by the forestry industry should have increased by 2020 by approximately 200 000 hectares to a total of 1 450 000 hectares of forest land in areas that are, or may develop into, high nature value areas. It is expected that the protection of forests with a high nature value will likely contribute to increasing national levels of biodiversity and recreation being supplied, although how this protection will affect the provision of other forest services at the regional and national scales is somewhat uncertain.

The study of future changes in socio-ecological systems is often done through simulation models that allow the representation of system dynamics across time and space, and their consequences under predefined scenario conditions (Schluter et al. 2012). Integrated assessment models combine diverse system elements across the boundaries of sectors, disciplines and system components (Hamilton et al. 2015; Harfoot et al. 2014). This kind of model provide a means to explore the connections and feedbacks between different system components, including the social, economic and ecological implications of different natural or anthropogenic factors. Integrated assessment models are often used to study land-use change (i.e. land-use models), its causes and its consequences.

Land-use change is the consequence of complex land-use decisions and results from multiple interactions between biophysical and socio-economic factors (Foley et al. 2005; Ojima et al. 1994). Behavioural and cognitive factors (e.g. objectives) have proven to have a strong influence on forest owner choices for management practices (Andersson and Gong 2010; Ingemarson et al. 2006). Silvicultural decisions are complex, largely due to the uncertainty associated with long time horizons in forest management (Blennow et al. 2014). Therefore, the behaviour and decision-making of forest owners should be incorporated as a determinant of land use and as a driver of land-use change in socio-ecological system models. The need to represent human behaviour and decision-making processes in models of complex socio-ecological systems is increasingly recognised (Dearing et al. 2010), and it has drawn in recent years the attention of the land-use modelling community towards agent-based models (ABMs) (Matthews et al. 2007). These models essentially consist of a number of 'agents' (i.e. entities with autonomous behaviour) that interact with each other and their environment and that make decisions as a result of these interactions (Ferber 1999). Within the context of socio-ecological systems, an ABM is made up of a population of agents, generally land managers, and a landscape within which they can act and interact (Rounsevell et al. 2012).

ABMs have developed considerably since they first came out in the 1970s (Hare and Deadman 2004), growing in complexity (Janssen and Ostrom 2006; Valbuena et al. 2010) and becoming increasingly popular in the social sciences and land system science (Matthews et al. 2007; Parker et al. 2003; Valbuena et al. 2010). Nevertheless, conceptual gaps and challenges still remain to accurately and empirically ground the representation of human behavioural processes and their links to the land system. To start with, ABMs have so far focused on representing individual decision-making at local or regional scales. However, no such models have been developed at larger scales (e.g. national, continental, global) (Rounsevell et al. 2012). Given that land uses and ES are influenced by many national and international policies, and that on-going global change debates are increasingly played out at national, continental and global scales, large-scale ABMs may be particularly informative. Nevertheless, such models are still difficult to construct, mainly because they are very data-demanding, but also due to the high computational power required to run simulations at such large scales at a resolution at which decision-making land units (e.g. landholding, forest stand) can be adequately represented.

A shortcoming of land-use/cover change models in general is the common assumption that land uses are uni-functional, being allocated to the production of a single good or service (e.g. meat, cereal, timber, recreation). In real-world systems, however, the majority of land uses generate multiple goods and services (Foley et al. 2005). Such multi-functionality is increasingly encouraged by national and international policies (Otte et al. 2007), and its widespread adoption is a crucial land use issue. Additionally, land-use intensity affects the production of goods and services as well as associated terrestrial and aquatic ecosystems (Socolow 1999; Tilman 1999). Even so, few land-use/cover change models have made an attempt at incorporating multi-functional land uses (Groot et al. 2009) or land-use intensity gradients (Renwick et al. 2013; Temme and Verburg 2011; Van Asselen and Verburg 2013), and only in the agricultural sector.

Institutions (e.g. government administrations, NGOs, society), which have a considerable influence on land managers through policy instruments and direct interventions, have generally been incorporated within these models in an indirect way through the creation of policy scenarios (e.g. Guzy et al. 2008; Ralha et al. 2013; Van Berkel and Verburg 2012). This approach can be useful in studies of disparities between the outcomes of various policies. In other cases, however, where the role of institutions is to be included in the model, but where scenarios are focused on other phenomena (e.g. effects of climate change, management strategies), the inclusion of institutions in the form of agents could make the model a more accurate representation of reality. Besides, the fact that institutions are created by people with their inherent behavioural mechanism of decision-making further justifies the need to attribute agency to them. Nevertheless, to date only three studies have incorporated institutional agents in some way in ABMs, two of which explored forest management strategies (Campo et al. 2009; Purnomo et al. 2005; Wang et al. 2013). Institutional agents would allow behaviour to be attributed to institutional entities, the inclusion of interactions between institutions and with other agents, and explicit treatment of the effects of such interactions and the environmental context on institutional policy decisions.

Models have great potential to explore forest socio-ecological systems, but their contribution has been limited in practice because of the various omitted system elements and drivers. Hence, there is a need to advance socio-ecological system models by incorporating such elements within them. To study the dynamics of forest socio-ecological systems I develop an ABM to address the abovementioned shortcomings of previous ABMs in this field with the purpose of enhancing model representativeness of real-world processes, using the case study of Sweden. I apply this model to the exploration of changes in socio-ecological systems associated with forestry, and the adaptation of this sector to environmental and climatic change. The expected rapid change in forest land use and management in Sweden due to regional and global environmental change effects justifies the need for an integrated study of land-use dynamics that accounts for both environmental change and decision-making from all involved stakeholders. Moreover, the underrepresentation of the interactions between the effects of land-use change and climate change in Swedish advisory and policy documents (Schroter et al. 2005; Ulmanen et al. 2012) further corroborates the need for such a study. Institutional interactions are addressed in a conceptual framework. Given that no formal model currently exists to assist the representation in ABMs of the decision-making, actions and interactions of institutions involved in forestry, I developed a conceptual model of these institutional processes.

Using the developed ABM I assess the consequences of different socio-economic and environmental scenarios in order to explore answers to the research questions (stated below) under a range of plausible future conditions. The Intergovernmental Panel on Climate Change (IPCC) offers a definition of the term scenario as used in the natural sciences (IPCC 2013):

"A scenario is a coherent, internally consistent and plausible description of a possible future state of the world. It is not a forecast; rather, each scenario is one alternative image of how the future can unfold."

Scenario development is performed using the formal scenario framework developed by (van Vuuren et al. 2014), which is meant to assist the generation of new scenarios to support research and assessment of adaptation and mitigation strategies and climate impacts. The scenario framework essentially combines the Shared Socio-economic Pathways (SSPs) (O'Neill et al. 2014) (i.e. socio-economic scenarios integrated within a space of challenges to climate change adaptation and mitigation) and Representative Concentration Pathways (RCPs) (van Vuuren et al. 2011) (i.e. radiative forcing pathways resulting from different greenhouse gas atmospheric concentrations) (Fig. 2). All scenarios will be defined at the national scale and will be projected up to the year 2100. The reason for modelling such a long period is that changes in forest management tend to take a long time to happen (i.e. several decades) due to the fact that trees need to grow for several decades before they reach optimal size for harvesting. Also, the relatively slow rate of climatic change and subsequent impacts on capitals (IPCC 2014b) justifies the long-term time horizon.

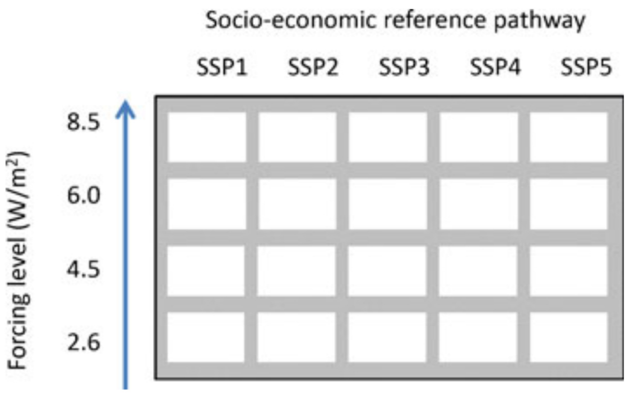


Fig. 2 Proposed scenario framework for adaptation assessment on the basis of the SSPs (columns) and the RCPs (rows) (van Vuuren et al. 2014)

With the aim of exploring uncertainty about how to adapt to future global change in Swedish forestry, and to improve modelling approaches to explore land-use change, ES provision and adaptation in the forestry sector, throughout this thesis I address four research questions, each of which is the focus of a thesis chapter:

1. What owner types and management strategies exist in the Swedish forestry sector and how are their management decisions made? (Chapter 2)

2. How might global change influence future land use change and ES provision in Sweden? (Chapter 3)

3. How can the forestry sector adapt to environmental change in meeting future demands for ES in Sweden? (Chapter 4)

4. How can institutions, their actions and interactions in the forestry sector be modelled? (Chapter 5)

Finally, findings from all four chapters, limitations of the study, future research, and policy relevance are jointly discussed in a final Thesis Discussion.

Chapter 2

Characterising forest managers through their objectives, attributes and management strategies

Adapted from Blanco, V., Brown, C., Rounsevell, M.D.A. 2015. *Characterising forest managers through their objectives, attributes and management strategies*. European Journal of Forest Research, 134:1027-1041.

1 Introduction

Forest land use and management has changed considerably in recent decades (Meyfroidt et al. 2010; Rudel et al. 2005; Siry et al. 2005) with globalization being identified as one of the main drivers of forest land use change (Meyfroidt et al. 2010; Seppala 2008). A shift in the industrial production of timber away from boreal and temperate forests to fast-growing tropical and subtropical forests has taken place since the 1980s, as trade between these areas increases (Seppala 2008), and as demand for timber products grows in many developing and newly industrialised countries, especially in the Asia-Pacific region (FAO 2012). This trend coupled with the increasing adoption of sustainable forest management and forest certification schemes, especially in developed countries (Seppala 2008; Siry et al. 2005), has supported the provision of non-timber goods and services such as recreation or biodiversity conservation (Seppala 2008). At the same time, the use of plantations to meet global demands for wood and fibre for industrial use has increased since the 1960s (FAO 2000; FAO 2005; Sohngen et al. 1999).

Furthermore, social and economic change in developed countries has led to the environmental, biological and recreational benefits of forestry becoming better recognised and valued (Janse and Ottitsch 2005), leading to increased demand for non-timber forest services. In particular, forest multi-functionality, understood as the capacity of a forest to provide multiple market and non-market ecosystem services (ES) (Millennium Ecosystem Assessment 2005; Ninan and Inoue 2013; Richnau et al. 2013), is encouraged increasingly by national and international policies (Otte et al. 2007), and its widespread adoption is a crucial land use issue.

In the midst of these global and macroeconomic drivers of forest land use change, forest managers make decisions about the management of their forests, and the subsequent provision of ES generated from forest land. Forest managers' attitudes towards forests and forestry, and the objectives for their forests, are perhaps the most important elements affecting management decisions (Ní Dhubháin et al. 2007; Nordlund and Westin 2011) and are likely to have substantial impacts on the range of goods and services provided (Arano and Munn 2006; Sorice et al. 2014; Urquhart and Courtney 2011). Hence, there is a need to investigate forest manager decision-making and its consequences for forest socio-ecological systems in order to inform land-use and forestry policy (Beach et al. 2005; Ingemarson et al.

2006; Urquhart and Courtney 2011) and sustainable forest management plans (Emtage et al. 2007; Wiersum et al. 2005). Additionally, such information can inform the development of simulation models as a means of representing interactions and feedbacks between 'agents' (i.e. entities with autonomous behaviour depicting real-world actors) and their environment; a priority for the provision of improved insight and understanding of socio-ecological systems (Ferber 1999; Rounsevell et al. 2012).

Despite the key role of forest managers in determining the supply of forest ES at the epicentre of global forest land-use change, no attempt has been made so far to characterise forest managers at global scales. Such large-scale studies need to recognise that each forest manager has their own unique characteristics and circumstances, rendering attempts to fully account for individual behaviour infeasible (Emtage et al. 2007). To deal with this heterogeneity within forest manager communities a common approach is to group together similar 'types' of land managers and then to detail the profiles of these groups. This leads to a land manager typology, which, whilst not describing individuals, depicts archetypal patterns that tend to repeat themselves within the community (Emtage et al. 2007). Hence, heterogeneity is reduced by creating clusters of land manager types, within which managers are expected to display somewhat similar behaviour and decision-making compared to individuals in other groups.

The creation of typologies is common in analyses of the agriculture sector (e.g. Guillem et al. 2012; Karali et al. 2013), but less so for forestry, where typologies have almost exclusively targeted specific local or national-scale cases. Such approaches have not been applied across scales where they could improve understanding of the management of forest systems internationally and the resulting provision of ecosystem goods and services, especially under global trade and environmental change.

Nevertheless, previous typological studies and reviews have suggested that a small number of broad classes may be sufficient to describe forest managers across large geographical scales (e.g. Beach et al. 2005; Ní Dhubháin et al. 2007; Wiersum et al. 2005).

Wiersum et al. (2005) observed that the management characteristics of forest owners[1] were statistically more commonly associated with countries than with types of rural area (these

[1] Throughout the text I refer to forest *managers* wherever studies refer to those responsible for management (owners or otherwise). I do occasionally use the term *owners* when referring to specific

differing in socio-economic and land-use characteristics). This suggests that, at very large geographical scales, large scale characteristics (e.g. national policies, culture) might better explain the forms of forest management practised than specific small-scale characteristics. Hence, I postulate that despite geographical heterogeneity, it should be possible to create forest owner typologies at large scales (i.e. supranational). These typologies may or may not replicate the same patterns found at lower scales (e.g. local, landscape), yet they can in principle depict the different types of owners according to the relative similarities and dissimilarities existing at the large scale.

This idea is supported by the Agent Functional Type approach to the development of agent typologies in the context of large-scale socio-ecological systems (Arneth et al. 2014; Rounsevell et al. 2012), which suggests that three dimensions be used in the definition of agent typologies: functional roles, agent desires or goals and behavioural mechanisms, with the second and third dimensions nested within the first. An agent type's overall 'function' in a socio-ecological system can therefore be denoted by functional roles such as *environmentalist* or *multifunctional* (as in the study of Wiersum et al. 2005). If a number of individuals within a forest manager community have similar attributes across the three dimensions, they can be represented by a single forest manager type. Similarities in attributes may increase or decrease across spatial scales.

Given the global and interconnected nature of drivers of forest land-use change, there is a clear need for forest manager typologies to be developed at supranational scales that can aid the understanding of forest manager choices and their implications at an international level. An international forest manager typology may be further used to create communities of agents that can populate agent-based models operating at global scales. These models could provide an understanding of land-use processes and socio-environmental interactions unprecedented at global scales, as no agent-based models have yet been created at such a scale (Arneth et al. 2014). Moreover, an international typology in conjunction with smaller scale nested typologies (e.g. national, local) may contribute a robust basis on which to construct forest policy and sustainable management plans (Emtage et al. 2007; Rounsevell et al. 2012).

---

studies that have used this term to maintain consistency with those studies; or when addressing issues directly related to forest owners that may not necessarily apply to managers (e.g. bequest).

To improve understanding and modelling of forest managers at international scales, I develop a qualitative forest manager typology based on a meta-analysis of quantitative and qualitative information about forest manager types and their decision-making strategies. I assess (1) whether groups of forest managers share characteristics across different locations and scales; and if so, (2) what these characteristics are and how they vary between groups; and (3) what forest manager functional types exist at the broad scale. Using this information I discuss a forest manager typology across gradients of environmental, social and economic benefits provided by forests, and within a sustainability framework. I further discuss the typology's implications for forest multi-functionality and for future research on land use decision-making and natural resource management.

2 Methods

I conducted a meta-analysis of the existing literature on forest manager and forest owner typologies and decision-making mechanisms (Fig. 3). Using the search database 'Web of Science' I searched, under the categories 'Topic' and 'Title', for the term combinations: *forest manager typolog**, *forest owner typolog**, *forest owner typ**, *forest manager typ**, *forest owner profile*, *forest manager profile*, *forest owner objective**, *forest manager objective**, *forest owner decision*, *forest manager decision*. For all publications in the search output lists I screened the title and abstract first, and, if there was a direct relationship to the topic of the study, I subsequently analysed the full paper. If these papers mentioned other, pertinent papers that were not identified during the initial search, these were included in the analysis. I restricted the analysis to papers published after 1990 to ensure that the information was up-to-date, while still covering a long period of time (24 years).

I selected 31 publications containing information directly relevant to the generation of a generic forest manager typology (Table A.1). Such information referred to forest managers' (principally private forest owners) values, attitudes, beliefs, objectives, decision-making mechanisms, socio-demographic and economic attributes, and management strategies. These studies covered different geographical scales and locations within Europe and the United States (Table 1). No information was found for macro-regions other than these.

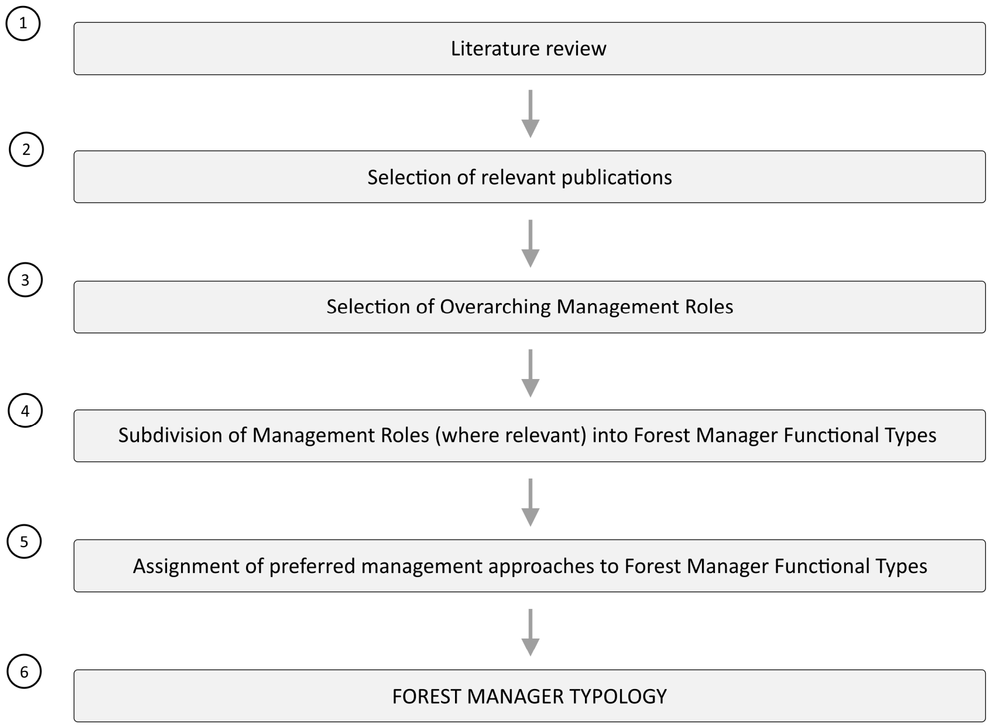


Fig. 3 Steps of the literature meta-analysis performed to develop the forest manager typology

Table 1 Number of publications cited per country and geographical scale at which the survey was conducted. Review papers and papers not relating to a particular geographical location were included under the category "Others"

	Geographic Scale				
	Subnational	National/ State(s) (US)	Supranational	Others	Total
Country/ Country cluster					
Sweden	2	5			7
Denmark		1			1
Finland	1				1
United Kingdom	1				1
Austria	1	1			2
Portugal		1			1
United States (1-5 states)	2	8			10
European Union (8 countries)			1		1
Others				7	7
Total	7	16	1	7	**31**

I used the Agent Functional Type conceptualisation to establish the structure of the agent types (Arneth et al. 2014; Rounsevell et al. 2012). This hierarchical structure incorporates the overarching management roles at the highest level, with subdivisions of these leading to manager functional types at the lowest level. Overarching management roles were selected from among recurrent forest manager types found in the literature (i.e. emerging in at least 5 papers) that relate to their management strategies and objectives. On occasion, different studies applied different names to types with very similar underlying characteristics and overall motivations (e.g. productionist, economist, and investor) according to the descriptions of types and the quantitative/ qualitative information behind them. In such cases these types were included under one overarching management role with shared characteristics. Where the internal variability of group characteristics was large, some overarching management roles were then subdivided into types. These types were defined according to subgroups found in the literature, which could be delimited within an overarching role because of their distinctive objectives and/or socio-demographic or economic attributes. The typology included those objectives and attributes that were either referred to as (at least) *somewhat/moderately important* in defining a forest manager type in at least three papers, or as *very important* at national/state scale or larger in at least one paper.

Based on the comprehensive classification of forest management approaches developed by Duncker et al. (2012a), I linked forest manager types with their management preferences (i.e. approaches). The choice of a particular management approach is based on decisions about the type of operation to implement during the development of a forest or stand (Duncker et al. 2012a). These decisions were defined through the following variables: naturalness of tree species composition, tree improvement, type of regeneration, successional elements, machine operation, soil cultivation, fertilization or liming, application of chemical agents, integration of nature protection, tree removals, final harvest system and maturity. I linked agent functional types with corresponding management approaches by considering the similarity in the content and coherence between functional types and possible management categories, and descriptions of management practices in the papers. I do not go into the details of particular operational decisions associated with each approach; for this information the reader is referred to Duncker et al. (2012a).

Each forest manager type was defined according to primary and potential secondary management/ownership objectives, ranges of manager socio-demographic and economic attributes, and preferred forest management practices. Resulting groups of this subdivision of the forest manager population were 'forest manager functional types'.

I then characterised the different forest manager functional types within the triple bottom line sustainability framework (Elkington 1994) by quantifying the environmental, social and economic impacts of each type and its associated management practices. An overall sustainability index was then determined by calculating the average value of the quantified impacts. To quantify the environmental impact I scored the five possible levels of management intensity and three possible levels corresponding to the importance of nature conservation and environmental quality objectives for the manager type. The social impact was quantified by scoring the importance given to objectives that could provide public services, namely public recreation, aesthetics, nature conservation, environmental quality and hunting. The economic impact was quantified according to the three levels of focus on profit-making objectives.

Score values for management intensity levels ranged from 0 to 0.4 and values for the importance of objectives ranged from 0 to 0.2, as shown in Table 2. These (arbitrary, but consistent) values were assigned following a semi-quantification of the objectives, attributes and management preferences of each manager functional type. The index generated for each impact was then the sum of the attributed values of the different characteristics corresponding to each manager type, divided by the sum of the maximum possible values of those characteristics. In this way, levels of management intensity and the importance of objectives, measured at different scales, were normalised (i.e. calculated on a common scale). Finally, the overall sustainability index, calculated by averaging the three impact index scores, took values between 0 (low sustainability) and 1 (high sustainability). While these indices are not intended to reflect meaningful absolute values, and the functional relationships between manager type characteristics and their degree of sustainability may not necessarily be linear, they do allow relative ranking of manager types.

I considered this to be the best method with imperfect and semi-quantified information and therefore, even though the overall index provides continuous numerical values, it is only meant to be a broad guide to sustainability.

The sustainability of each functional type was located within a three-circle Venn diagram, commonly used to depict the triple bottom line framework, using values of the environmental, social and economic indices for each type on the corresponding axes. An equilateral triangle with corners at the furthest point of each circle from the centre of the diagram defined these axes, which spanned values 0 to 1. The point representing the sustainability of each functional type within the diagram was the centroid of a triangle with corners on the positions of the three corresponding index values.

Table 2 Score values assigned to the different levels of management intensity and to the importance of objectives used to generate the forest manager functional type sustainability index

Score values	0	0.05	0.1	0.15	0.2	0.25	0.3	0.35	0.4
Management Intensity	Intensive	Intensive -High	High	High-Medium	Medium	Medium -Low	Low	Low-Passive	Passive
Objectives	Not Important		Secon dary		Primary				

3 Results

I found that management types and practices aligned along gradients of the provision of private versus public goods and services, generation of profit versus non-profit goods and services, and management intensity (Fig. 4). I define public goods and services as those ES provided by private or public land from which the general public may benefit, whether they are delivered on-site (e.g. recreation, aesthetic pleasure) or off-site (e.g. water purification, carbon sequestration). The profit versus non-profit gradient follows a general trend in the economic focus of forest manager types from those whose only objective is to maximise economic profit from forest activities, to those who have little or no interest in profit-making. Both the private/public goods and services and the intensity gradients follow a similar trend in terms of the positions of types along them. These gradients reflect management approaches selected by forest managers according to their objectives (Duncker et al. 2012a) and socio-economic attributes. Generally, more profit-oriented managers are found to be willing to manage their forests more intensely and so occur at one extreme of each gradient.

Sustainability index scores varied substantially between manager types, although none approached the most extreme values possible under the scoring method (Table 3). Scores for environmental and social impacts were, for most owner types, relatively close to each other, suggesting a possible correction between both sets of scores. This may be due to the fact that nature conservation and environmental quality objectives were accounted for in the calculation of both the environmental and social impact scores, as these objectives carry substantial relative weight in the calculation of both scores. Fig. 5 illustrates the relative positions of the functional types within the sustainability framework. I note that the different types span a larger range along the economic axis than they do along the environmental and social axes.

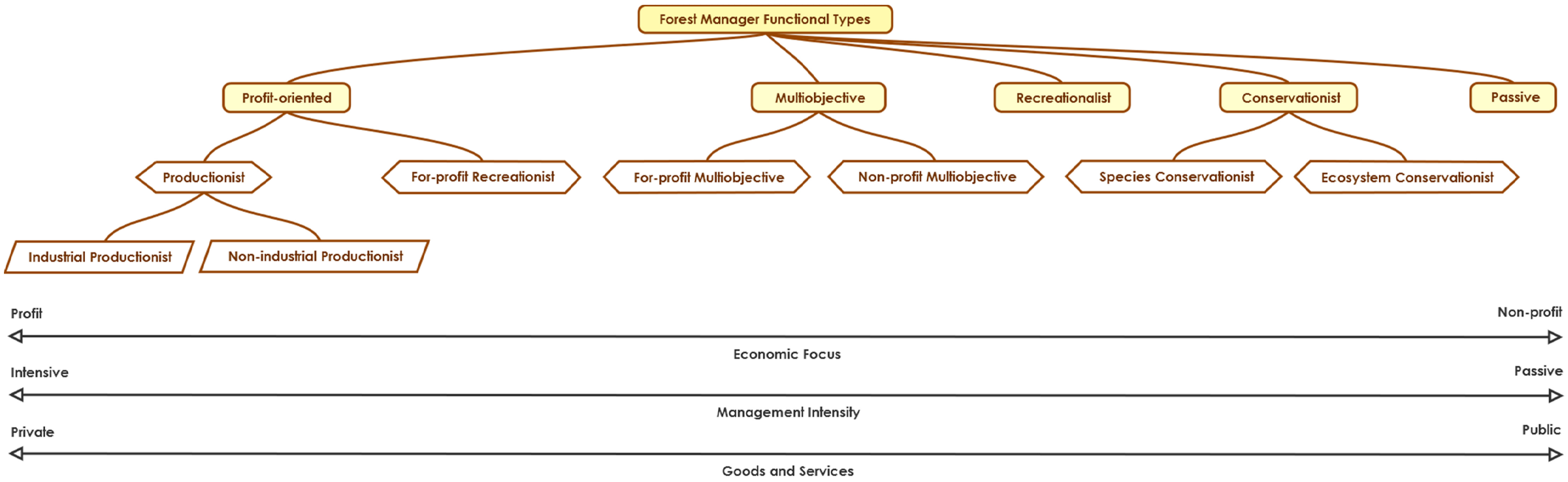


Fig. 4 Forest manager types separated into functional groups and their approximate relative positions on axes describing the economic focus and intensity of their management, and the nature of the goods and services they produce. Nine Agent Functional Types were identified

Table 3 Index values (0-1) calculated for each forest manager functional type according to their capacity to fulfil the environmental, social and economic dimensions of sustainability, and the sustainability index resulting from the averaging of values for these dimensions for each type. I assume equal importance for the three dimensions

	Environmental	Social	Economic	Sustainability
Industrial Productionist	0.06	0.00	1.00	0.35
Non-industrial Productionist	0.31	0.40	1.00	0.57
Profit-oriented Recreationist	0.43	0.40	1.00	0.61
For-profit Multi-objective	0.50	0.40	1.00	0.63
Non-profit Multi-objective	0.87	0.80	0.50	0.72
Recreationalist	0.69	0.60	0.00	0.43
Conservationist	0.94	0.70	0.50	0.71
Passive	0.75	0.30	0.50	0.52

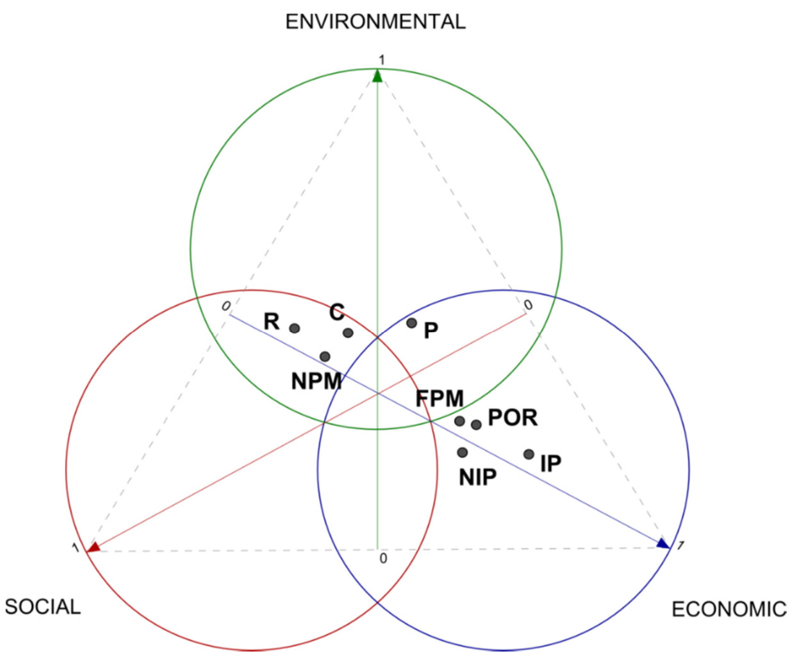


Fig. 5 Conceptualisation of forest manager functional types within the triple bottom line sustainability framework. The location of each type is a function of its position along the environmental, social and economic gradients determined by their corresponding index values (Table 3) for Industrial Productionist (IP), Non-Industrial Productionist (NIP), Profit-Oriented Recreationist (POR), For-Profit Multi-objective (FPM), Non-Profit Multi-objective (NPM), Recreationalist (R), Conservationist (C) and Passive (P)

I find ten different objectives to be somewhat important in determining the forest management preferences of one or more forest manager functional types (Table 4). Additionally, eight socio-demographic or economic attributes are found to determine the definition of one or more forest manager functional types.

Table 4 Dimensions of the forest manager functional typology (in bold) and the different attributes within each dimension that a forest manager must or may have

OVERARCHING MANAGEMENT ROLE	**MANAGEMENT OBJECTIVES**	**SOCIO-DEMOGRAPHIC/ ECONOMIC ATTRIBUTES**	**FOREST MANAGEMENT PREFERENCES**
Profit-oriented	Profit-making	Age	Management intensity
Multi-objective	Private consumption	Educational level	
Recreationalist	Personal enjoyment	Forestry knowledge	
Conservationist	Public recreation	Gender	
Passive owner	Aesthetics	Income	
	Nature conservation	Property size	
	Environmental quality	Location of residence	
	Cultural conservation	Property acquisition	
	Hunting	Possession of Forest Management Plan	
	Privacy		

Our analysis resulted in five overarching forest management roles: profit-oriented, multi-objective, recreationalist, conservationist, and passive owner (Fig. 4). Subdivisions of some of these roles produced nine forest manager functional types: industrial productionist, non-industrial productionist, for-profit recreationist, for-profit multi-objective, non-profit multi-objective, recreationalist, species conservationist, ecosystem conservationist and passive owner. I provide narrative descriptions here of all forest manager types that are included in the typology in terms of their main objectives, socio-demographic/ economic attributes and forest management preferences. These narratives start with the overarching management role and follow the functional type hierarchy, where pertinent, down to individual forest manager functional types. Table 5 shows all characteristics defining each type, including those not described in the narratives.

Some trends are found to apply to all forest managers. Possibly due to bequest considerations, older owners are less likely to engage in harvesting or in wildlife and recreation improvement activities (Joshi and Arano 2009). Other general findings are that the types interested in ecosystem management tend to have higher education levels (Creighton et al. 2002), and that female owners tend to have more pro-environmental, recreational and human-centred values and attitudes (Nordlund and Westin 2011; Stern et al. 1993).

3.1 Profit-oriented

Objectives

The main objective of profit-oriented managers is profit-making. Within this group three forest manager functional types can be distinguished, which I call 'industrial productionists', 'non-industrial productionists' and 'for-profit recreationists', with the first two representing the majority of the profit-oriented group. There is a general consensus within the productionist group about the high importance of timber production and forest ownership as an investment (Boon et al. 2004; Ingemarson et al. 2006; Kline et al. 2000b; Majumdar et al. 2008).

Socio-demographic and economic attributes

Productionists are found to have, in general, lower levels of education than conservationists or passive owners, but higher levels than multi-objective owners (Ingemarson et al. 2006). Nevertheless, productionists have less forestry knowledge (i.e. knowledge about forest management) than multi-objective owners. Generally, owners with a higher income are less likely to engage in harvesting (Joshi and Arano 2009), which I interpret as showing less dependency on income from forestry. Indeed, a large proportion of productionists have low forest income dependency, although a significant fraction of them have medium or high dependency (Canadas and Novais 2014; Ingemarson et al. 2006).

Table 5 Semi-quantification of Forest Manager Functional Types (FMFT) according to their primary (✓✓) or secondary (✓) objectives and socio-demographic/ economic attributes, and forest management preferences. Names of manager types in bold refer to overarching management roles, while names not in bold to the right of each overarching role are the FMFTs comprised within that role. A manager type may cover more than one forest income dependency category: Low (L), Medium (M), and High (H), and may be made by a high (H) or a low (L) proportion of residents (R) or absentees (A). Educational level, forestry knowledge and property size are categorised for each manager type in relative terms (Lower, Medium (Med.), Higher) with respect to the other types

	PROFIT-ORIENTED	PRODUCTIONIST	INDUSTRIAL PRODUCTIONIST	NON-INDUSTRIAL PRODUCTIONIST	FOR-PROFIT RECREATIONIST	**MULTI-OBJECTIVE**	FOR-PROFIT MULTI-OBJECTIVE	NON-PROFIT MULTI-OBJECTIVE	**RECREATIONALIST**	**CONSERVATIONIST**	SPECIES CONSERVATIONIST	ECOSYSTEM CONSERVATIONIST	**PASSIVE**
Objectives													
Profit-making	✓✓	✓✓	✓✓	✓✓	✓✓	✓✓	✓✓	✓		✓			✓
Private Consumption						✓	✓	✓	✓	✓			
Personal Enjoyment						✓	✓	✓	✓✓	✓			
Public Recreation					✓✓				✓				
Aesthetics	✓	✓		✓	✓	✓✓	✓	✓✓	✓✓	✓✓			✓
Nature Conservation	✓	✓		✓		✓✓	✓	✓✓	✓	✓✓			✓
Environmental Quality	✓	✓		✓		✓✓	✓	✓✓	✓	✓✓			✓
Cultural Conservation	✓	✓		✓		✓✓	✓	✓✓	✓	✓			
Hunting	✓	✓		✓	✓	✓✓	✓	✓✓	✓	✓			
Privacy						✓	✓	✓	✓	✓			
Attributes													
Age	✓					✓			✓	✓			
Educational Level		Lower ✓				Lower ✓			Higher ✓	Higher ✓			
Forestry Knowledge		Med. ✓				Higher ✓							Lower ✓
Gender	✓					✓			✓	✓			✓
Income Dependency		L, M, H ✓✓				L, M, H ✓✓	L, M, H ✓✓	L, M ✓✓	L, M ✓✓	L ✓✓			L, M ✓✓

Property Size			Much larger ✓✓	Larger ✓✓		Larger ✓✓	Larger ✓✓	Med. ✓✓	Smaller ✓✓	Smaller ✓✓			Smaller ✓✓
Location of Residence		R: H A: L ✓✓				R: H A: L ✓✓			R: L A: H ✓✓	R: L A: H ✓✓			R: L A: H ✓
Property Acquisition	✓					✓				✓			
Forest Mgmt. Plan		✓				✓				✓			✓
Mgmt. Prefs.													
Management Intensity			Intensive, High	Intensive, High	Low, Passive		Medium	Low	Low, Passive		Low, Passive	Low, Passive	Passive

Productionists tend to own much larger properties than recreationalists, passive or “non-timber” owners (Boon et al. 2004; Canadas and Novais 2014; Eggers et al. 2014; Karppinen 1998; Kline et al. 2000b; Majumdar et al. 2008), probably because of their interest in maximising forest income (Arano and Munn 2006). Resident owners tend to have stronger productionist values and stronger economic management attitudes (Nordlund and Westin 2011), and Ingemarson et al. (2006) found a larger proportion of productionists living on or near their estate than conservationists or passive owners. Finally, productionists are more likely to have a forest management plan than any other owner type (Eggers et al. 2014; Ingemarson et al. 2006; Majumdar et al. 2008).

Management preferences

Productionists are more likely to carry out intensive forest management and to use single species plantations than any other type of owner (Arano and Munn 2006; Duncker et al. 2012a; Fujimori 2001).

3.1.1 Profit-oriented forest manager functional types

Within the productionist group, two manager functional types can be distinguished: industrial and non-industrial productionists. Unlike non-industrial private forest owners, industrial forest owners generally own and operate a commercial wood processing plant and manage forests almost solely for timber and biomass production on the basis of profit maximization (Arano and Munn 2006; Beach et al. 2005; Liao and Zhang 2008; Newman and Wear 1993). Industrial productionists also manage far larger properties than non-industrial productionists and generally manage them more intensely. They both fall within the "intensive" or "high" intensity classes of Duncker et al. (2012a).

The for-profit recreationist type comprises owners who intend to make a business out of recreation associated with nature, adventure and outdoor sports activities, or hunting (Andersson *pers comm*. 2014; Matilainen and Lahdesmaki 2014) rather than timber. Their main objectives are likely to be profit-making and recreation, while they also give importance to aesthetics. Those making businesses out of hunting also attribute importance to game production. In Sweden, this functional type makes up a very small proportion of productionists. For-profit recreationists are expected to manage their forests in a non-intensive way, and differently depending on their recreational focus. They may fall within either the "passive" or "low" intensity classes of Duncker et al. (2012a).

3.2. Multi-objective

<u>Objectives</u>

Multi-objective owners are characterised by attributing high importance to several objectives. Like the productionists, they see forest ownership as an investment and concentrate on timber production (Ingemarson et al. 2006; Kline et al. 2000b; Majumdar et al. 2008). In Sweden, annual income was also seen as important (Ingemarson et al. 2006). Personal enjoyment in the form of recreation, mushroom and berry picking or appreciation of green space is also regarded as an important objective by multi-objective owners. In the UK, they also valued public recreation (Urquhart and Courtney 2011). Other objectives prioritised by this class include aesthetics, game management and hunting, nature conservation and environmental quality (the latter including water and soil conservation, climate change mitigation and pollution control) (Ingemarson et al. 2006; Majumdar et al. 2008; Urquhart and Courtney 2011). In Sweden, multi-objective owners also value cultural conservation very highly (Ingemarson et al. 2006).

Socio-demographic and economic attributes

Multi-objective owners generally have lower education levels than conservationists, productionists or passive owners, and yet they have higher average forestry knowledge than any of these groups (Ingemarson et al. 2006). Like productionists, multi-objective owners with greater experience in the forestry business seem to be prepared to take relatively large risks (Andersson 2012). In Sweden, the proportion of female owners was lower amongst multiobjectivists than amongst productionists, conservationists or passive owners. While a large proportion of multi-objective owners have low or medium dependency on their forest income, a significant minority displayed high dependency (Ingemarson et al. 2006). As with productionists, multi-objective owners with higher incomes are less likely to engage in harvesting (Joshi and Arano 2009). They generally have much larger properties than recreationalist, passive (Kline et al. 2000b) and "non-timber" owners (Majumdar et al. 2008). A larger proportion of multi-objective owners than either conservationists or passive owners live on or near their estate (Ingemarson et al. 2006).

Management preferences

Multi-objective owners can be expected to manage forests with more than one tree species. Hence, they manage either a mixed forest or several fragments of different forest types. They also implement extended rotation periods (i.e. beyond the optimum economic harvest age) in order to allow for the biodiversity benefits created by older forests (Kline et al. 2000b).

3.2.1 Multi-objective forest manager functional types

Multi-objective managers give relatively high importance to several different and potentially competing objectives, and there is often a large variability among the managers in this group in the relative importance they give to their objectives. Hence, they may be subdivided into smaller clusters depending on the relative emphasis they put on particular objectives. I draw the main subdivisions by looking at the two predominant groups of forest owners that Ní Dhubháin et al. (2007) observed. The primary objective of the first group was the production of wood and non-wood goods and services, usually for profit, while the second group's main objective was the consumption of such goods and services. I call these groups *for-profit multi-*

objective managers and *non-profit multi-objective* managers respectively, as the main difference between them lies in the importance they give to profit-making objectives. Hence, the for-profit multi-objective functional type places larger importance on profit-making objectives relative to the other main objectives of multi-objective owners. The non-profit multi-objective type, conversely, prioritises every other objective over the profit-making objective. These managers are comparable with Majumdar et al. (2008) “non-timber” owners and Ross-Davis and Broussard (2007) “new forest owners”.

For-profit multi-objective and non-profit multi-objective managers are expected to differ mainly in their forest income dependency and the size of their properties. Because non-profit multi-objective managers do not prioritise profit-making as highly as for-profit multi-objective, their forest income dependency is likely to be low or medium, while that of for-profit multi-objective managers is expected to span a wider range, including highly dependent multi-objective managers. At the same time, assuming that the profit-making objectives of for-profit multi-objective managers are similar to those of productionists, they are expected to own larger estates than non-profit managers. For-profit multi-objective managers generally fall within the “medium” intensity class of Duncker et al. (2012a) while non-profit managers perform “low” intensity management.

3.3 Recreationalist

Objectives

Recreationalists’ primary objectives are personal enjoyment and aesthetics (Boon et al. 2004; Kline et al. 2000b; Majumdar et al. 2008), and often informal public recreation (e.g. walking, cycling, cross-country skiing, nature watching) (Urquhart and Courtney 2011). A substantial proportion of this group also judge nature conservation and environmental quality, hunting, private consumption of timber and fuel wood, cultural conservation and privacy to be important (Boon et al. 2004; Ingemarson et al. 2006; Majumdar et al. 2008; Urquhart and Courtney 2011); Andersson *pers comm*. 2014).

Socio-demographic and economic attributes

There is a tendency for recreationalists to have higher education levels than productionists, multi-objective or passive owners (Kline et al. 2000b). This could be because recreationalists also tend to have higher non-forest incomes, as income can be partly explained by formal education level (Griliches and Mason 1972). Additionally, it is unlikely that recreationalists will have high income dependency on forests, given that they generally use their forests for their own enjoyment. Recreationalists very often own much smaller properties than productionists or multi-objective owners (Boon et al. 2004; Joshi and Arano 2009; Karppinen 1998; Kline et al. 2000b; Majumdar et al. 2008) and are commonly absentee owners (Karppinen 1998).

Management preferences

Recreationalists are likely to own natural forests and forests largely comprising broadleaf deciduous trees, as these are generally perceived as more aesthetically pleasing than coniferous forests (Fujimori 2001). Forests with several successional stages (i.e. different stand development stages) also seem to contribute to this perception. Recreationalists fall within either the “passive” or “low” intensity classes of Duncker et al. (2012a).

3.4 Conservationist

Objectives

Conservationists’ primary objective is nature conservation, followed by aesthetics and environmental quality (Ingemarson et al. 2006; Majumdar et al. 2008; Urquhart and Courtney 2011). An appreciable number of managers in this group also value cultural conservation, timber production, private consumption, hunting, personal enjoyment and privacy (Ingemarson et al. 2006; Majumdar et al. 2008; Urquhart and Courtney 2011); Andersson *pers comm*. 2014).

Socio-demographic and economic attributes

There is a tendency for the conservationist manager group to include a larger proportion of females than the productionist or the multi-objective groups (Ingemarson et al. 2006). Also, conservationists often have higher education levels and lower income dependencies than productionist, multi-objective or passive owners (Creighton et al. 2002; Ingemarson et al. 2006). In Sweden, the vast majority of conservationists had low forest income dependency, while only a very small proportion had medium dependency. Conservationists also tend to own much smaller properties than productionists or multi-objective owners (Eggers et al. 2014; Majumdar et al. 2008), and usually live further away from their forest than these and passive owners (Ingemarson et al. 2006; Nordlund and Westin 2011). In Sweden, while retaining the property within the family is considered a principal goal by most owners (Lidestav 2010; Lönnstedt 2012), conservationists had the highest proportion of owners who had bought, rather than inherited, property (Ingemarson et al. 2006).

Management preferences

Conservationists are likely to own mixed, natural or old growth forests with several successional stages and native species (Fujimori 2001). They commonly practice extensive – or no – management and allow natural growth. Those conservationists with an interest in timber production will practice extended rotation periods (Kline et al. 2000b)(Kline et al. 2000). Conservationists fall within either the “passive” or “low” intensity classes of Duncker et al. (2012a).

3.4.1 Conservationist forest manager functional types

In terms of nature conservation goals, two main conservationist management strategies could be distinguished: species conservation and ecosystem conservation. Conservation of small or declining populations seeks to prevent particular species from becoming locally and/or globally extinct (Caughley 1994), while the ecosystem approach to conservation aims to preserve biodiversity and ecosystem functions rather than single species (Franklin 1993). Hence, forest management differs depending on the conservation goal. Population conservation is likely to entail more intensive management as the forest system may have to be moulded to cater to the needs of one or a few species (Baker et al. 2011), while ecosystem conservation will often imply lower management intensity.

3.5 Passive

Objectives

Passive owners typically do not give high importance to any particular objective and have low or no engagement in the management of their forests (Boon et al. 2004; Ingemarson et al. 2006). However, the fact that some Swedish passive owners had medium forest income dependency and that 33.5% of them had a forest management plan no older than 10 years (Ingemarson et al. 2006) suggests that some do have profit-making objectives.

Socio-demographic and economic attributes

Passive owners have been recorded as having the lowest forestry knowledge of any owner group (Ingemarson et al. 2006). The majority of them have low forest income dependency, while a small proportion has medium dependency (Eggers et al. 2014; Ingemarson et al. 2006). They also tend to own much smaller properties and to live further away from these than productionists and multi-objective owners, yet closer than conservationists (Eggers et al. 2014; Ingemarson et al. 2006; Kline et al. 2000b).

Management preferences

Passive owners generally do not manage their land. Those with profit-making objectives may undertake the minimal required management to make some profit from their forests. In general, passive owners fall within the “passive” management intensity class of (Duncker et al. 2012a).

4 Discussion and conclusions

I present a generic typology of forest managers that goes beyond the continental scale, including forest manager types found across a number of developed countries spanning Mediterranean, warm-temperate, nemoral, continental and boreal biomes. By analysing previously-published qualitative and quantitative information about forest managers, I identified a small number of manager types for which there was consistent evidence across the developed world. I observed that similar types were found at the level of the overarching management-role by studies performed at different locations and scales (e.g. Karppinen 1998; Majumdar et al. 2008; Urquhart and Courtney 2011).

This does not mean that I expect the typology to hold strictly as presented here for every location and at every scale within the developed world. However, it does summarise the forest manager community in these areas at a 'global' scale and can serve as a starting point in studies of these managers at particular locations. While most or all forest manager functional types may be expected to be present in developed countries and administrative regions with considerable forest cover, the proportion of managers falling within each of these functional types will vary from place to place. Furthermore, there will be within-functional-type variability in particular attributes between different manager communities. Our use of classes (e.g. Low, Medium, High) as opposed to continuous values to subdivide attributes reflects the uncertainty about these attributes.

It should be noted that the coverage of forest managers in this study is largely limited to private forest owners, while other types of managers such as local communities, indigenous people, NGOs or religious organisations (e.g. the church) were not included. These types of managers may differ from private forest owners in their objectives and attributes, but I believe that the overarching roles I identify are likely to hold for at least some of these other types.

It is also important to note that a series of somewhat arbitrary choices were made in the development of this typology that may have affected the final outcome. First, the choice of terms and term combinations, and the cut-off year used in the literature search determined the publications that went into the review. Had these choices been different, it is possible that other relevant articles could have come up. The selection of objectives and attributes included in the typology (given by a minimum number of papers that an objective/attribute had to be mentioned in as (at least) *moderately* to *very important*) was also arbitrary to some degree. Finally, the choice of objectives and number of objectives considered for each pillar (i.e. type of impact) of the sustainability framework, and the number of levels scored for each objective, were restricted by the categories found in the typology. If objectives and levels included in the framework had been chosen on a different basis, sustainability index scores might have been somewhat different.

The different forest manager functional types in the typology can be associated with three gradients according to (1) their economic focus, (2) the intensity of management associated with their objectives and (3) the type of goods and services they provide. The profit vs. non-profit gradient concurs with the dichotomy highlighted by Beach et al. (2005), who distinguished profit-maximisers from utility-maximisers. Our typology further arranges manager functional types according to the degree of importance that they place on profit-making and non-pecuniary utility generation within their objectives. Awareness of the particular economic foci of, and the objectives pursued by, different manager types found at a location can help to determine the type of policy instruments to be put into effect. For instance, while profit-oriented managers tend to be motivated by financial instruments (i.e. economic incentives and disincentives), recreationalists or conservationists, having little interest and dependency on profit generation through their forests, are likely to be more influenced by information and advisory services that can instruct them on issues such as nature restoration or biodiversity conservation (Boon et al. 2004; Ingemarson et al. 2006).

The forest manager functional types can also be separated along an intensity gradient. Having coupled the functional types with the five forest management approaches proposed by Duncker et al. (2012a), the typology of forest managers follows a similar trend in management intensity as in the classification of their forest management approaches. I interpret management intensity as the degree of manipulation of natural processes (Duncker et al. 2012a), and this broad definition allows us to qualify the intensity of management not only for production purposes, but also for a number of other objectives (e.g. recreation, conservation), which may involve very different management practices and intensities.

Partly as a result of the approaches taken to their management, forests generate various ES. Public institutions are increasingly encouraging private forest owners to provide public-good benefits (Boon et al. 2004; Ingemarson et al. 2006; Kline et al. 2000b; Urquhart and Courtney 2011). As we follow the gradient in goods and services provision from industrial productionists to passive managers, there is a general increase in the proportion of public goods provided and a decrease in private goods. Commensurate with this finding, previous studies have observed that ecological and societal goals are prioritised in unmanaged and "close to nature" forests (e.g. Duncker et al. 2012a; Gamfeldt et al. 2013; Ninan and Inoue 2013). This tendency is especially strong in regions (such as Sweden or Slovenia) where private forests are open to the general public (Eriksson 2012; Ficko and Boncina 2013). Where forests are closed to the public, access to public goods and services such as recreation or aesthetics is clearly limited (Finley and Kittredge 2006; Urquhart and Courtney 2011). However, the fact that forests provide services such as water purification beyond their boundaries may render this gradient true even for private forests without public access. The relevance of this gradient depends therefore on the nature of the services provided by a particular forest and on where these services are delivered.

Sustainable forest management recognises the necessity of balancing the ecological, social and economic outputs from forests (MCPFE 2003). However, it can be difficult to ascertain what degree of sustainability can be expected in managed forests given the wide range of managerial objectives, forest types and management practices. I illustrate the relationship between the different manager functional types in terms of their sustainability by placing them within the triple bottom line framework. From this conceptualisation it appears that multi-objective and conservationist managers are generally the most sustainable types, as might be expected given the large number of objectives they manage for. In contrast, industrial productionists emerge as the least sustainable managers given their almost exclusively economic focus, followed by recreationalists, penalised for attributing no importance to economic objectives. Even so, the fact that a generic functional type may hold some variability within its objectives, attributes or management strategies implies that the values taken by the index may consequently vary for each type within a certain spectrum. Therefore, sustainability index values generated here should be taken with caution and understood as approximate for the generic manager types.

This conceptualisation links well with the concept of multifunctional land use, which attempts to maximise the diversity of goods and services that a land unit can provide. Multi-objective managers are an obvious example of land users aiming for multifunctionality, and this largely ensures that they achieve high sustainability scores compared to industrial productionists, for instance. However, multifunctionality can potentially be addressed at different scales, through multifunctional landscapes and even regions. It has been argued that at these scales, a range of specialised, often mono-functional, land uses within an area can provide a multiplicity of ES (Vereijken 2002; Wiggering et al. 2006). In such cases, more specialised manager types such as productionists or conservationists may have larger roles. It may be bold however to assume that, for the same land area, a combination of specialised land-uses and a multifunctional land use will be able to supply the same amounts of the same ES and that these will be distributed spatially in a similar fashion (Le Du-Blayo 2011). The approach used may in the end be determined by local and regional conditions (Cocklin et al. 2006). A sustainability index as presented here that scores manager functional types may not be sufficient to evaluate sustainability at the landscape or regional level, whereas an index that scores the sustainability of different combinations of functional types could be useful for this purpose.

While it would be desirable to develop a typology that covers the different types of forest managers found across the globe, the typology presented here does not account for the developing world due primarily to a lack of relevant literature. A forest manager typology for developing countries is in principle likely to differ substantially from the one presented here. While in the developed world the environmental and recreational elements of forestry have become more important in recent decades as a result of social and economic developments (Janse and Ottitsch 2005; Nordlund and Westin 2011), the focus in most developing nations remains on forest utilization for income generation and subsistence (Arnold and Perez 2001; Seppala 2008). Therefore, production-oriented management will likely dominate in these countries, while management for recreation and conservation is likely to be much less common. Furthermore, it may not be possible for profit-oriented managers in low income economies to implement high-intensity management practices due to a lack of access to the necessary infrastructure and financial capital. Also, the use of forest products for personal consumption in subsistence communities may be considerable, while it is rare in the developed world (Urquhart and Courtney 2011).

Networks and knowledge transfer are an additional key element to consider when studying forest manager interactions and decision making (Beratan 2007). The way managers interact, who they interact with and the degrees of trust with which they interact strongly affect how they deal with complexity and uncertainty in the land-use system. It has been observed here for instance that a large proportion of forest managers are absentee owners and are therefore likely to seek information in different ways and places than managers residing on or near their property. While more traditional social networks among resident land managers may not apply to absentees, their interactions with forestry cooperatives, the forest administration, and other forest advisors near a place of residence could be crucial. In the case of industrial productionists, information exchange may even occur at trans-regional or trans-national scales, making accurate representation of networks very important within the context of a globalised forestry sector. Further studies are needed to explore how different types of forest managers interact and gather information relevant to their forests.

An additional important consideration is that typologies may evolve over time (Emtage et al. 2007). The typology presented here represents a snapshot of the forest manager community over a particular period of time. Despite the typology incorporating studies of forest managers across 24 years, it does not reflect the "evolutionary trajectory" (Landais 1998; Paquette and Domon 1999) of the different manager types across this period. To reduce the uncertainty associated with trying to understand future land-use with 'time-point' typologies, research on the ways in which manager types evolve, learn and adapt to environmental change is required.

Further in-depth studies are also needed to construct a qualitative global typology covering all existing forest manager types. As developing countries are absent from our typology and the forest manager typology literature in general, future research should aim to fill this knowledge gap. Despite the above caveats, the fact that a typology of forest managers can be clearly distinguished from the literature and aligned along gradients of management focus, intensity, motivation and sustainability, suggests that it is both possible and useful to develop global typologies of forest managers and land managers in general. These could assist policy making by supporting policies that are orientated toward functional types and the development of resource management programmes and agent-based models of land use processes at international scales. The incorporation of such a typology within an agent-based model that includes a way of representing land manager decision-making and behavioural processes could support studies of future land use change at large scales. Insights from such studies can in turn be valuable for land-use policy planners to inform international policy (e.g. conventions, directives).

Chapter 3

The effect of forest owner decision-making, climatic change and societal demands on land-use change and ecosystem service provision in Sweden

---

This chapter has been submitted for publication to the journal *Ecosystem Services*. The forest manager typology developed in Chapter 2 was applied in the development of the model described in the present chapter.

1 Introduction

Land-use and land management change have important effects on the provision of ecosystem services (ES) (Millennium Ecosystem Assessment 2005). Forests provide a wide range of essential ES, including timber and non-timber products, air purification, carbon sequestration, biodiversity preservation and recreation, which make fundamental contributions to human societies and natural systems (De Groot et al. 2010; Millennium Ecosystem Assessment 2005). Concerns about greenhouse gas emissions as well as future shortages and rising prices of fossil fuels have led to a growing interest in wood biomass as a renewable energy source (Buonocore et al. 2012; Zanchi et al. 2012). Future agricultural demands and climatic change will also likely impact the distribution of forests and their levels of ES provision (Alexandratos and Bruinsma 2012; Schroter et al. 2005; Soja et al. 2007; Tilman et al. 2001). Hence, forest management strategies are being revised (e.g. Jonsson et al. 2015; Kjaer et al. 2014) and future land-use change assessed (e.g.Thompson et al. 2011) in order to allow adaptation to changing conditions and to meet future demands for ES supply.

Demands for ES are, however, often difficult to estimate (Hayha et al. 2015). Therefore, ES assessment often maps supply through the assessment of suitability (e.g. Hayha et al. 2015; Sohel et al. 2015) or vulnerability (e.g. Metzger et al. 2008; Tzilivakis et al. 2015), which need not consider ES demand. Mapping of ES supply is also performed through ES valuation (e.g. Costanza et al. 1997), which assumes demands non-explicitly. Furthermore, where ES demands are acknowledged, only demands for services with a market value are actually quantified (e.g. Verkerk et al. 2014). As a result, no study has measured the provision of non-marketable ES in relation to demand levels; a necessary step in understanding the conditions and changes needed to fulfil societal needs for ES.

Haines-Young and Potschin (2013) define ES benefits as the final outputs from ecosystems that have transformed into products or experiences that are not functionally connected to the systems from which they were derived. Yet, again, as a benefit for something can only be conceived if there is a demand for it, the quantification of societal benefits from particular services is dependent on their demand levels. Hence, it is not possible to understand how ES provision equates to ES benefits without assessing ES demands.

Land-use and land management change, which strongly determine ES provision, are ultimately dependent on the decisions of land owners. These decisions are driven by owner objectives and attitudes, which are often diverse and complex, but which allow for some categorisation of owners in order to consider possible present and future activities and their consequences (Chapter 2; Karali et al. 2013). Such categorisations are particularly useful in agent-based modelling (ABM) of land-use change, allowing the decision-making of individual land managers to be simulated efficiently and across large geographical extents (Matthews et al. 2007; Valbuena et al. 2010).

The strong influence of behavioural and cognitive factors (e.g. objectives) on forest owner choices for management practices (Andersson and Gong 2010; Ingemarson et al. 2006; Vulturius et al. in review) justifies the use of ABM to explore land-use change. The adoption of these models to map the effects of human behaviour on ES is however recent (Boone et al. 2011; Brown et al. 2014; Murray-Rust et al. 2011). Given the complex nature of silvicultural decisions (largely imposed by the uncertainty associated with long time horizons in forest management) (Blennow et al. 2014) they have seldom been incorporated into ABMs, and then only modelled up to the landscape scale (Rammer and Seidl 2015). These complexities also make it very difficult to simulate ES provision in forestry, because this provision depends so strongly upon individual decision-making. Therefore, a clear need exists for developments in simulation methods that allow for the links between ES provision and forest management change to be explored.

The need for improved modelling of the forest sector is best illustrated by countries such as Sweden, which have large forest areas that are economically and culturally important, and which are likely to be affected by climatic change. A 69% forest cover (SLU 2015) of which approximately 50% is owned by individual owners (Swedish Forest Agency 2015) with diverse objectives, and the fact that in 2011 forestry accounted for 2.2% of GDP showcase the importance of forestry in Sweden. Hence, under the highly uncertain future faced by forest ES, reliable ways of exploring future land-use change at large scales are needed. Future uncertainty is commonly explored through scenario analysis. To develop consistent scenarios of future climate and global change, van Vuuren et al. (2014) developed a framework that combines the Shared Socio-economic Pathways (SSPs) (O'Neill et al. 2014) (i.e. socio-economic scenarios integrated within a space of challenges to climate change adaptation and mitigation) and Representative Concentration Pathways (RCPs) (van Vuuren et al. 2011) (i.e.

radiative forcing pathways resulting from different greenhouse gas atmospheric concentrations). Hence, I developed an ABM that accounts for land owner decision-making and that is capable of appraising future ES provision in the context of ES demands, applied to future changes under the SSP-RCP scenarios from 2010 until 2100. The purpose of this exercise was to explore: a) future ES provision and how ES demands may be met, b) land-use change, and c) changes in land owner objectives, in Sweden.

2 Methods

To explore future ES provision and land-use change in Sweden, focusing on forestry, I developed the CRAFTY-Sweden model, based on the CRAFTY-CoBRA agent-based modelling framework, which in turn is an extension of CRAFTY (Competition for Resources between Agent Functional Types) (Murray-Rust et al., 2014) (see Appendix B for the model ODD protocol). CRAFTY allows the representation of large-scale land-use dynamics, based on demand and supply of ES (e.g. timber, food). Demand is given exogenously while supply depends on the productivities and behaviours of modelled agents, and the productivities of agents' locations (described by capitals representing the availability of resources such as infrastructure, human capital and crop suitability). Geographical space is represented as a grid of cells, each of which has defined levels of a range of capitals. Each cell may be managed by a single land-use agent, which uses the capital stock available within the cell to provide services according to its own production function. The competitiveness of a given level of service provision can be calculated on the basis of societal demands, overall supply levels and 'benefit' functions, which describe the monetary and non-monetary value to society of service production. Agents can make decisions based on their current competitiveness and participate in an allocation procedure with potential new agents that may result in land-use change. I use agent functional types (Rounsevell et al. 2012) (hereafter agent types) for the definition of agent production and behaviour. This approach helps to characterise agent typologies that define general characteristics of agents, from which individual agents can subsequently be drawn.

From here on, all model components and developments described are my own work unless otherwise stated.

2.1 Model description

In CRAFTY-Sweden, agents include different types of forest owners and farmers. Farmers were defined to simulate the competition for land between forestry and agriculture. Forest owner decision-making involves four key components: 1) owner objectives and associated management practices, 2) the time of felling, 3) an estimation of the future benefits agents expect to obtain from their land-use, 4) and their willingness to abandon, change management or hand over land to a different owner considering their competitiveness. Farmers consider all but the second component. Using land productivities and infrastructure, modelled forest owners are able to produce timber from different tree species, carbon sequestration, biodiversity, and recreation, while modelled farmers choose to produce one or more services among cereal, meat and recreation (Fig. 6).

In the following, I describe the development of the land owner typology, and the owner production and decision-making mechanisms. I also explain how baseline capitals, land-use and land owner types were mapped throughout Sweden. Finally, I describe the approach to scenarios and the analysis of simulation results.

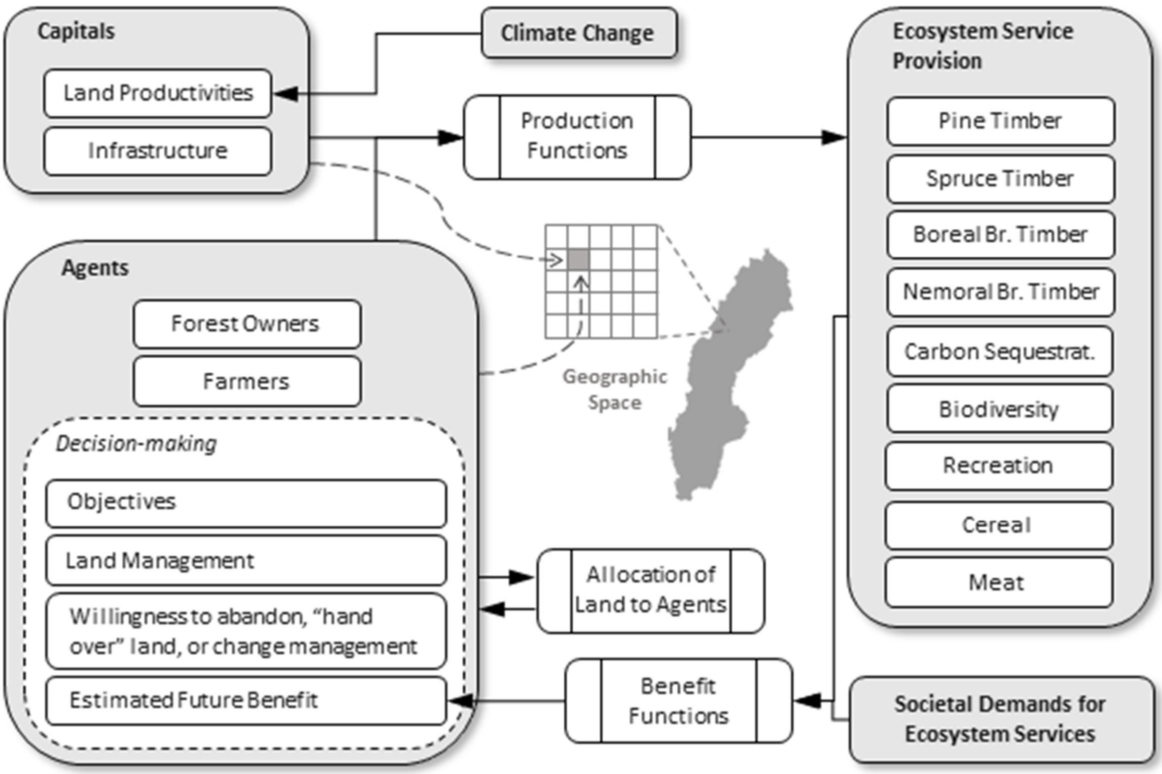


Fig. 6 *Schematic representation of the structure of the CRAFTY-Sweden model showing flows (solid lines) and associations (dashed lines) between components*

2.1.1 Land owner typology: design and validation

I developed a typology of forest and agricultural agents, focusing especially on the former. To define forest owner types I used as a basis the forest owner typology described in Chapter 2. Because this typology was based on studies performed at different scales and contexts to those found in Sweden, I performed a validation exercise of the typology using empirical information from 872 Swedish forest owners (Vulturius et al. in review). A cluster analysis showed that the five overarching management roles identified by the theoretical typology (productionist, multi-objective, recreationalist, conservationist and passive) were also clearly discernible in the empirical data. The cluster analysis was performed by Gregor Vulturius, from the Stockholm Environment Institute. Supplementary materials on this validation can be found in Appendix C.1.

Within each overarching management role, different options for forest management are possible, including the use of different types of forest (defined by species composition). Forest types were assigned to each management role on the basis of existing forest stand compositions (Swedish Forest Agency 2015) and potential adaptation measures to climate change that consider species composition, number of thinnings and rotation lengths (Felton et al. 2016; Jonsson et al. 2015) on the basis of owner objectives (Chapter 2; Duncker et al. 2012a). Forest types assigned were pine (*Pinus sylvestris*), spruce (*Picea abies*), boreal broadleaf (*Betula pendula, B. pubescens, Alnus incana, A. glutinosa, Populus tremula*), nemoral broadleaf (*Fagus sylvatica, Quercus robur, Fraxinus excelsior, Ulmus glabra, Tilia cordata, Carpinus betula*), and combinations of these, resulting in 17 forest owner types. The management and decision-making strategies of each owner type are described in sections 2.1.4, 2.1.5, and 2.1.6.

Given the current levels of agricultural production (Swedish Board of Agriculture 2009) and management intensities prevailing in Sweden (Institute of Environmental Studies 2015), farmers were separated by the main services provided (i.e. cereal or meat) in combination with their main objectives (i.e. commercial or non-commercial).

2.1.2 Capitals

The capitals that agents can use in service production are productivities for pine, spruce, boreal broadleaf, and nemoral broadleaf forests, grassland productivity (natural capital), and transportation infrastructure (infrastructure capital). Table 6 shows capital descriptions, their data sources, and the ES they contribute to producing (see Appendix C.2 for further detail on the calculations that led to final capitals).

Table 6 Identities and data sources for modelled capitals, and the ecosystem services they contribute to producing

CAPITAL	DEFINITION	INPUT DATA (units; resolution)	ECOSYSTEM SERVICES	DATA SOURCE
Pine, Spruce, Boreal Broadleaf, and Nemoral Broadleaf Forest Productivities	Baseline productive potential for each forest type	Forest production potential per forest type (m^3sk ha^{-1} yr^{-1}; $1km^2$)	- Pine, spruce, boreal br. and nemoral br. timber - Carbon - Biodiversity	(Hägglund and Lundmark 1987) (Johansson et al. 2013) SLU
Grassland Productivity	Baseline productive potential for grassland and cropland	LPJ-GUESS simulated C3-grass NPP projection for 2010 driven by climate (radiation, temperature, precipitation) (kg C m^{-2} yr^{-1}; 50x50 km)	- Meat - Cereal	Simulations done for this project by Dr. Fredrik Lagergren (see section 2.2)
Transportation Infrastructure	Proximity to transportation networks and central markets	1. Road and rail networks 2. Waterway networks 3. Travel time to nearest town with over 50000 inhabitants ($1km^2$)	- Pine, spruce, boreal br. and nemoral br. timber - Meat - Cereal - Recreation	1. UNECE 2. EEA 3. GEMU, JRC

SLU: SLU Forest Map, produced by Swedish University of Agricultural Sciences, accessed via ftp://salix.slu.se/download/skogskarta
UNECE: United Nations Economic Commission for Europe, accessed via http://www.unece.org/trans/areas-of-work/transport-statistics/statistics-and-data-online.html
EEA: European Environment Agency, accessed via http://www.eea.europa.eu/data-and-maps
GEMU, JRC: Global Environment Monitoring Unit, managed by the Joint Research Centre, accessed via http://www.edenextdata.com/?q=content/jrc-accessibility-map-estimated-travel-time-nearest-city-population-50000

2.1.3 Baseline land-use and land owner distribution

Land-use map

To create a baseline land ownership map for 2010 I first devised a land-use map at $1km^2$ resolution that included pine, spruce, pine-spruce, pine-boreal broadleaf, spruce-boreal broadleaf, boreal broadleaf, and nemoral broadleaf productive forests, agriculture, protected areas, non-productive forests, semi-natural vegetation, wetlands, open spaces, ‘other unmanaged’ land, artificial, and water bodies. SLU Forest Map data (SLU 2010) on the proportion of different tree species per cell were used to identify forest cover and classify it according to the forest types assigned above, according to the proportion of forest within the cell, and the proportions of different species within that forest. CORINE land cover (EEA 2014) was used to identify all other land use/land cover classes. Nationally Designated Areas (EEA 2015) were then superimposed to define protected areas. Non-productive forests are also protected and unavailable for production (Swedish Forest Agency 2014c). Thus, I identified them by:

1. Assigning to forested cells the value of the highest productivity found among all forest types within that cell; and
2. Given the proportion of non-productive forest per county (Swedish Forest Agency 2015), selecting for each county the equivalent number of cells with the lowest productivity values.

Mean forest age values from the SLU Forest Map were used to assign forest ages.

Agent locations

Forest owner types were allocated to productive forest types using data about: a) the area of productive forest land by county and ownership classes for 2010 (Swedish Forest Agency 2015); and b) the proportion of owners in each county belonging to each group from the cluster analysis. Agricultural land and (some) semi-natural vegetation were assigned to commercial cereal, non-commercial cereal, commercial livestock, and non-commercial livestock farmer agents according to the land-use intensity in 2010 (Institute of Environmental Studies 2015). Remaining semi-natural vegetation, wetlands, and ‘other unmanaged’ land were left unallocated. Protected areas, non-productive forests, open spaces with little or no vegetation, artificial surfaces, and water bodies were not available for allocation during simulations. Fig. 7 shows the resulting map of Sweden. Further detail on the creation of land-use and land owner type maps can be found in Appendix C.3.

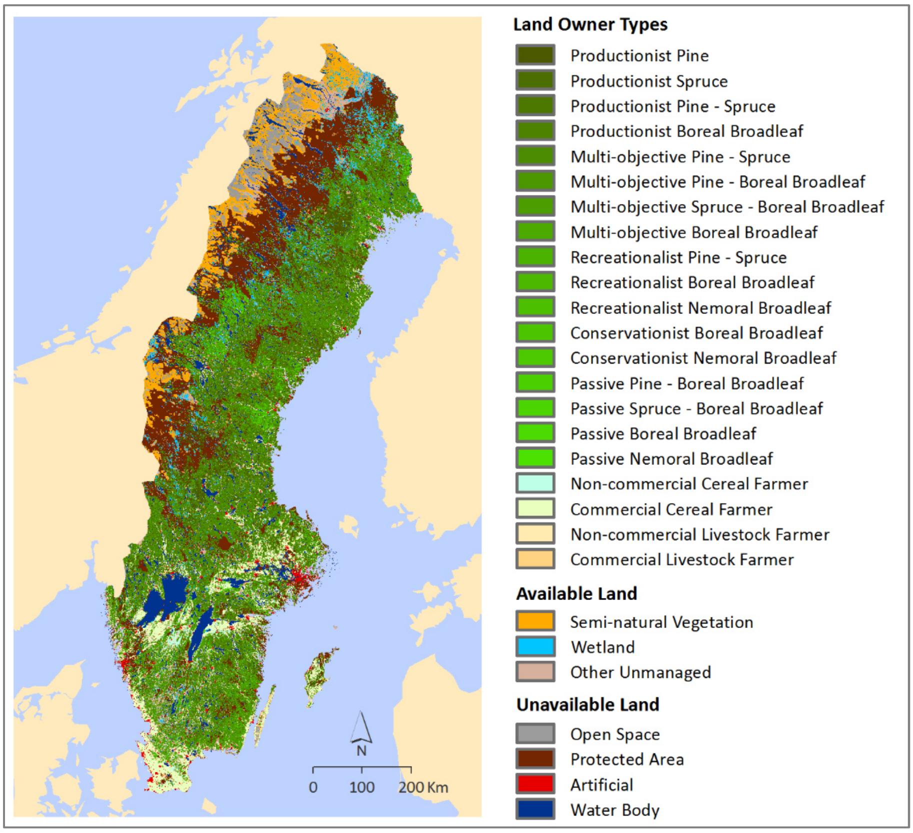


Fig. 7 Map of land owner type distribution throughout Sweden in 2010. Available and unavailable land refer to land that can or cannot be managed by agents, respectively

2.1.4 Land owner service production

The production of agricultural services was modelled on an annual basis. Forestry services are however dependent on forest age. Additionally, climatic change can affect service production by acting on productivities. I therefore developed ways of modelling the time-dependent component of the different services in CRAFTY. The production of a service by an agent in a given year was based on the Cobb Douglas function, adapted to incorporate a time component (Eq. 1).

$$p_s = o_{s,t} \prod_c \left(c_i + \Delta c_{i,t}\right)^{\lambda_c} \tag{1}$$

Production (p_s) of a service s within a cell is the product for all capitals (c) of the optimal production that an agent type would be able to achieve in a given year (o_{st}), and the unit-less (i.e. [0-1]) cell capitals (c_i) plus annual climate-induced change in cell capitals (Δc_{it}), weighted by the capital sensitivities of that agent type (λ_c) (Table 7). To reflect individual variability, optimal production o_{st} was uniformly randomly drawn from [$0.95\overline{o_{s,t}}, 1.05\overline{o_{s,t}}$], and capital sensitivity levels from [$\overline{\lambda_c} - 0.1, \overline{\lambda_c} + 0.1$]. Production calculations for each service are described below.

Timber

For timber production, o_{st} is given by a forest owner type-specific function that determines timber growth given the forest's age. The ProdMod model (Eko 1985) was used to generate timber growth curves for each owner type given their management preferences. Given passive owners' generalised lack of primary objectives for forestry, I assumed them to inherit forest, and therefore only enabled them to take over the forest and associated optimal production function of other owner types managing forests with the same tree species as them. Hence, optimal production functions were not calculated for passive owners. Table 7 shows parameter values used in ProdMod that differed for each owner type. See Appendix C.4.1 for further detail on optimal timber production function calculation.

Table 7 Land owner type production, felling age and competitiveness (scenario-independent) parameters. Number of stems planted per ha for each forest type, site index, number of thinnings implemented, age at each thinning per forest type, and percentage removed per thinning are parameters given to ProdMod to calculate the (age-dependent) optimal timber production and (above ground) carbon sequestration functions. These functions and remaining parameters in this table are CRAFTY-Sweden inputs. Yearly o_s illustrates yearly optimal farmer production. Productivity and infrastructure sensitivities (λ) are given per service. Felling age means (μ) and standard deviations (σ) represent number of years past minimum felling age (m.f.a.) of a forest, given as the m.f.a. range dependent on site quality

	Service Production								***Felling Age***		***Competitiveness***
Land Owner Type	**No. Stems/ha**	**Site Index cm**	**No. Thinnings**	**Age at each Thinning**	**% Removed per Thinning**	**Yearly o_s (tonnes)**	**λ Productiv.**	**λ Infrastr.**	**m.f.a. Range**	**μ, σ Years past m.f.a.**	**Probability Giving-up**
Productionist Pine	2500	280	3	24, 39, 54	25, 20, 20	-	0.8[a], 1[e], 0.06[f]	0.2[a], 1[g]	65-100	12, 10	0.05
Productionist Spruce	2600	360	3	23, 38, 53	25, 20, 20	-	0.8[b], 1[e], 0.06[f]	0.2[b], 1[g]	45-95	10, 8	0.05
Productionist Pine-Spruce	1250, 1300	280, 360	3	24/23, 39/38, 54/53	25, 20, 20	-	0.8[a,b], 1[e], 0.06[f]	0.2[a,b], 1[g]	45-95	10, 8	0.05
Productionist Boreal Br.	2200	320	3	15, 30, 45	25, 20, 20	-	0.8[c], 1[e], 0.06[f]	0.2[c], 1[g]	40-60	9, 7	0.05
Multi-objective Pine-Spruce	1150, 1250	280, 360	2	24/23, 39/38	30, 25	-	0.85[a,b], 1[e], 0.06[f]	0.1[a,b], 0.8[g]	45-95	15, 12	0.05
Multi-objective Pine-Boreal Br.	1840, 420	280, 320	2	24/20, 39/35	25/50, 20/25	-	0.85[a,c], 1[e], 0.06[f]	0.1[a,c], 0.8[g]	65-100	10, 8	0.05
Multi-objective Spruce-Boreal Br.	2000, 420	360, 320	2	23/20, 38/35	25/45, 20/25	-	0.85[b,c], 1[e], 0.06[f]	0.1[b,c], 0.8[g]	45-95	10, 8	0.05
Multi-objective Boreal Br.	2100	320	2	15, 30	30, 25	-	0.85[c], 1[e], 0.06[f]	0.1[c], 0.8[g]	40-60	15, 12	0.05
Recreationalist Pine-Spruce	1100, 1100	280, 360	3	24/23, 39/38, 54/53	25, 20, 20	-	0.9[a,b], 1[e], 0.06[f]	0.3[a,b], 0.6[g]	45-95	80, 14	0.05
Recreationalist Boreal Br.	2000	320	3	15, 30, 45	25, 20, 20	-	0.9[c], 1[e], 0.06[f]	0.3[c], 0.6[g]	40-60	100, 14	0.05

Recreationalist Nemoral Br.	1250, 1250	350, 300	3	25/22, 40/37, 55/52	25, 20, 20	-	0.9[d], 1[e], 0.06[f]	0.3[d], 0.6[g]	110-150	60, 14	0.05
Conservationist Boreal Br.	2100	320	1	15	35	-	0.9[c], 1[e], 0.06[f]	0.3[c], 0.8[g]	40-60	100, 14	0.05
Conservationist Nemoral Br.	1250, 1250	350, 300	1	25/22	35	-	0.9[d], 1[e], 0.06[f]	0.3[d], 0.8[g]	110-150	60, 14	0.05
Passive Pine-Boreal Br.	-	-	-	-	-	-	0.9[a,c], 1[e], 0.06[f]	0.1[a,c], 1[g]	65-100	25, 17	0.05
Passive Spruce-Boreal Br.	-	-	-	-	-	-	0.9[b,c], 1[e], 0.06[f]	0.1[b,c], 1[g]	45-95	25, 17	0.05
Passive Boreal Br.	-	-	-	-	-	-	0.9[c], 1[e], 0.06[f]	0.1[c], 1[g]	40-60	15, 10	0.05
Passive Nemoral Br.	-	-	-	-	-	-	0.9[d], 1[e], 0.06[f]	0.1[d], 1[g]	110-150	10, 10	0.05
Commercial Cereal	-	-	-	-	-	201	0.8[h]	0.5[h]	-	-	0.2
Non-commercial Cereal	-	-	-	-	-	121	0.5[h]	0.3[g,h]	-	-	0.2
Commercial Livestock	-	-	-	-	-	324	0.6[i]	0.5[i]	-	-	0.2
Non-commercial Livestock	-	-	-	-	-	193	0.3[i]	0.2[g,i]	-	-	0.2

[a] Pine timber, [b] spruce timber, [c] boreal broadleaf timber, [d] nemoral broadleaf timber, [e] carbon sequestration, [f] biodiversity, [g] recreation, [h] cereal, [i] meat

Carbon sequestration

Due to the difficulty of calculating soil carbon levels in interaction with forest productivities, only above-ground sequestered carbon (excluding the stump) was calculated. Optimal production functions of above ground carbon were also calculated using ProdMod outputs (Appendix C.4.2).

Biodiversity

The calculation of optimal forest biodiversity production considered forest age (Duncker et al. 2012b; Koskela et al. 2007; Marchetti 2004), using the generation of coarse woody debris with age as a proxy (e.g. Berg et al. 1994; Jonsell et al. 1998; Siitonen 2001), tree diversity (Gamfeldt et al. 2013; Marchetti 2004) and management practices undertaken by each owner type (e.g. woody debris removal), which have an influence on biodiversity (Chapter 2; Duncker et al. 2012a; Duncker et al. 2012b). I chose these forest attributes as indicators of biodiversity because of the availability of baseline data for them and the possibility of updating the data during model simulations. Finally, I considered the effect of forest productivity on biodiversity, specifically on coarse woody debris (Sturtevant et al. 1997), by assigning sensitivities to timber productivities. For further details of the calculation of optimal biodiversity production functions see Appendix C.4.3.

Recreation

Recreational value in Scandinavia is largely determined by the age of a forest, but also by forest management practices, accessibility and, to a lesser extent, by the types of tree species present (i.e. conifer vs broadleaf, and monoculture vs mixed) (Edwards et al 2012). See Appendix C.4.4 for further detail on optimal recreation function calculation.

Cereal and meat

Given baseline maps with available capitals and commercial cereal, non-commercial cereal, commercial livestock and non-commercial livestock agent locations (see section 2.1.6), their o_s and λ_c were adjusted until total cereal and meat production equalled the total production in Sweden reported by the FAO (2015) for 2010. The production of non-commercial agents was set at 0.6 times that of the commercial agents to reflect approximate differences in production potentials across equivalent classes in Van Asselen and Verburg (2013).

2.1.5 Forest Felling

The forest in a cell is clear-felled when it reaches an age that depends on site quality (i.e. productivity) (Lagergren et al. 2012) and owner objectives. In Sweden, the stand age at felling is regulated by law for pine and spruce to guarantee that the production potential is utilised (Kunskap Direkt 2015), and for beech, birch and oak recommended rotation periods exist (Löf et al. 2009; Rytter et al. 2008). Hence, lowest minimum felling age was assigned to the highest productivity values, while highest minimum felling age corresponded to the lowest productivity values (Table 7; Appendix C.5). Also, each owner type was assigned a Gaussian distribution of the planned felling age (above minimum felling age) (Table 7). This distribution was defined as being within the recommended rotation periods for all owner types except for recreationalists, conservationists and passive owners managing broadleaf forests. As these latter groups are not primarily interested in timber production (Chapter 2), they were assigned felling age distributions beyond the recommended rotation period. Felling age is determined at the time that an agent is allocated to a cell by randomly drawing a number (i.e. age) from within the agent type's distribution. Upon felling, timber is harvested and carbon sequestered in the standing timber is removed from the national pool.

2.1.6 Competition for land

Farmers can be taken over by other agents each year because they are assumed to manage on annual timescales. For forest owners however, I assume that they will not abandon or change the management approach on their land until the forest has reached maturity, in order to recover the initial investment. Hence, competition for forested land starts only once the minimum age of felling has been reached. At that point a 'potential' agent with a higher competitiveness score than the incumbent agent can take over its land, resulting in one of two outcomes:

1. If the potential agent is a forest owner type willing to plant the same forest type as that already standing in the cell, it will inherit the production functions of the former owner, as the effect of changing management of a forest once maturity is reached is negligible. Age of felling is however adjusted to meet the objectives of the new agent. As mentioned in section 2.1.3, passive owners follow this system exclusively and do not compete for unmanaged land.

2. If the potential agent is a farmer or a forester not meeting the above criteria, the standing forest is clear-felled and land is either converted to farmland or to newly-planted forest.

Forest owners plan what they will plant according to (non-climate sensitive) charts that show potential tree growth according to site conditions. Even though some owners may also consider climate change and risk spreading, this is currently not a generalizable trait of Swedish forest owner decision-making (Blennow et al. 2012). Hence, while farmer service production is evaluated for the coming year, forest owners evaluate it for the (future) year of felling. To evaluate agent competitiveness for a given bundle of services I use the benefit (β) function (Eq. 2):

$$\beta = \sum_{s} \frac{p_s/A_i}{d_s} \, b_s \, \frac{u_s}{d_s} \tag{2}$$

where the production level p_s is time-discounted by forest age at felling (A_i) to reflect desire for shorter-term returns where possible. Time-discounted production is normalised by the current per-cell demand (d_s) (i.e. demand divided by the total number of cells) to achieve levels that are comparable across agent types supplying services that are measured in different units. The per-cell unmet demand (u_s), is also normalised by the demand to give a proportional unmet demand. Finally, b_s is a weighting factor representing the importance of the unmet demand of each service. A_i and b_s are parameterised to reflect observed time discounting and the assumed importance to society of meeting service demand levels, respectively.

If an existing agent's competitiveness is lower than its 'giving-up' threshold, it will abandon the cell. If a potential agent's competitiveness within a cell is greater than the existing agent's by a value larger than the existing agent's 'giving-in' threshold, then the potential agent takes over the cell. Giving-up and giving-in thresholds reflect minimum acceptable benefit and tolerance to competition respectively, and are drawn from agent type-specific Gaussian probability distributions to simulate individual differences (Murray-Rust et al. 2014). Also, because not all farmers and foresters are affected by market conditions to the same degree or at the same time, I implement giving-up probabilities for each agent type that apply to agents whose competitiveness falls below their giving-up threshold. Giving-up and giving-in mean values are scenario-dependent and are given in section 2.2, while standard deviations were set at 0.1 for all agents. Giving-up probabilities are given in Table 7.

2.2 Scenario analysis

Five future scenarios were defined by combining RCPs and SSPs. RCP4.5 was combined with SSPs 1, 3 and 4, and RCP 8.5 with SSPs 3 and 5, so as to explore coherent combinations of emission and socio-economic futures (Carter et al. 2015). Each RCP was also simulated with three climate models. Each climate model-RCP combination consisted of a different set of climate-induced annual productivity changes. Table 8 presents scenario-specific parameters.

The ecosystem model LPJ-GUESS (Smith et al. 2001) was used to simulate forest dynamics during 2010-2100 using climate projections of the Global Circulation Model-Regional Circulation Model ensembles (hereupon ‘climate models’) EC-Earth-RCA4, IPSL-RCA4 and NorESM-RCA4 for RCPs 4.5 and 8.5 from the EURO-CORDEX project (Jacob et al. 2014; Jones et al. 2011). LPJ-GUESS simulations were performed by Dr. Fredrik Lagergren, from Lund University. Annual climate-induced change was calculated for all productivities using LPJ-GUESS spatial projections of yearly timber volume growth for pine, spruce, boreal broadleaf and nemoral broadleaf forests, and yearly net primary productivity (NPP) change for grass until 2100 at 50x50 km resolution. Upon checking for non-linearities in volume growth and NPP change projections, linear models were considered to be adequate. Therefore, a regression coefficient was calculated for every cell by performing linear regression on projected growth values. These values were then downscaled to 1 km^2. See Appendix C.6 for more details on calculations of climate impact on productivities.

Following land-use and European SSP storylines from Engström et al. (2016) and Kok et al. (2015) respectively, SSPs differed in: a) future demands for ES, b) probability distributions for owner type giving-in and giving-up thresholds, c) the importance to ‘society’ of meeting demands for each service, and d) the possibility of farmland displacing forest land. Baseline demands for timber, cereal and meat were assumed to equal the observed production in 2010 (FAO 2015; Swedish Forest Agency 2015), while those for carbon sequestration, biodiversity and recreation were assumed to equal the simulated baseline supply. Future projections were calculated using the IIASA SSP data (IIASA 2015) on decadal rates of change of global forest land cover (for timber and carbon sequestration), and crop and livestock demands. Demands for biodiversity were projected following the SSP storylines with guidance from modelled global future changes in species abundance from UNEP (2007). Rates of change in recreational demands were assumed to be the same as those for biodiversity. Giving-in thresholds were set higher and giving-up thresholds lower for SSPs with greater barriers to adaptation (i.e. SSPs 3 and 4). See Appendix C.7 for further details about the creation of service demand projections.

CRAFTY-Sweden simulations were run for Sweden for the 2010-2100 period at a $1km^2$ resolution. The model was calibrated to produce minimal short-term (decadal) changes in land management under constant levels of demand and productivities, so that the effects of long-term forest management and scenario conditions could be isolated. The model was then run under these static conditions for the period 2010-2100 to produce a reference scenario. To understand model behaviour, sensitivity analysis was performed by altering values of behavioural, benefit function components, demands and productivities individually. To measure the effect of random model components 32 simulations were run under different random seeds, but otherwise identical parameterisations. Consequently, each climate model-RCP-SSP combination was run once (under one random seed).

2.3 CRAFTY-Sweden outputs

Modelling outputs presented here are for land-use change and ES provision. Agent types were mapped for every year during 2010-2100, and are presented for each scenario grouped into land-use and functional role categories through maps depicting hotspots of change, and figures showing nationally and regionally aggregated change. Hotspots were defined as 50x50 km (smaller next to country borders and on islands, down to $39km^2$) units of analysis where the category with the highest proportional increase (calculated as the mean increase from the three climate model runs for the scenario divided by the area of the analysis unit) experiences an increment above 10%. Following the larger administrative divisions used by the Swedish Forest Agency, regional changes were aggregated into the Swedish regions: Upper Norrland, Lower Norrland, Svealand and Götaland, and shown as percentage changes between 2010 and 2100 (i.e. area change relative to available area in the region) with error bars showing the variability in the change between maximum and minimum values generated among the three climate models. Finally, ranges of service provision defined by maximum and minimum values among climate models were plotted for each scenario.

Table 8 Scenario matrix. Service demands are only shown for the years 2010, 2050 and 2100. Demands shown for 2010 under the Reference scenario also apply to all other scenarios

	Reference			SSP 1 - RCP 4.5		SSP 3 - RCP 4.5		SSP 3 - RCP 8.5		SSP 4 - RCP 4.5		SSP 5 - RCP 8.5	
Service Demands	**2010**	**2050**	**2100**	**2050**	**2100**	**2050**	**2100**	**2050**	**2100**	**2050**	**2100**	**2050**	**2100**
Pine Timber (mill. m^3sk)	35.29	35.29	35.29	37.98	40.22	32.85	31.14	33.00	31.77	36.71	36.24	34.97	35.30
Spruce Timber (mill. m^3sk)	44.45	44.45	44.45	47.84	50.66	41.37	49.22	41.56	40.01	46.23	45.65	44.04	44.46
Boreal Br. Timber (mill. m^3sk)	7.64	7.64	7.64	8.22	8.71	7.11	6.74	7.14	6.88	7.95	7.85	7.57	7.64
Nemoral Br. Timber (mill. m^3sk)	1.51	1.51	1.51	1.63	1.72	1.41	1.33	1.41	1.36	1.57	1.55	1.50	1.51
Carbon Sequestration (mill. ton)	592	592	592	637	675	551	522	554	533	616	608	587	592
Biodiversity (1000, unitless)	234	234	234	234	257	210	183	206	175	222	211	199	189
Recreation (1000, unitless)	345	345	345	345	380	311	270	304	258	328	312	293	279
Cereal (mill. ton)	4.32	4.32	4.32	6.28	6.27	6.60	8.46	6.55	8.38	5.82	5.88	7.12	7.30
Meat (1000 ton)	537	537	537	765	718	808	966	815	998	761	781	1226	1178
Importance of meeting service demands		$\boldsymbol{b_s}$		$\boldsymbol{b_s}$		$\boldsymbol{b_s}$		$\boldsymbol{b_s}$		$\boldsymbol{b_s}$		$\boldsymbol{b_s}$	
Pine Timber		1.1		1.1		1.1		1.1		1.09		1.1	
Spruce Timber		1.1		1.1		1.1		1.1		1.09		1.1	
Boreal Br. Timber		0.045		0.045		0.045		0.045		0.040		0.045	
Nemoral Br. Timber		0.010		0.010		0.010		0.010		0.007		0.01	
Carbon Sequestration		0.2		0.35		0.1		0.1		0.1		0.1	
Biodiversity		1.1		2.50		1.0		1.0		1.0		1.0	
Recreation		1.1		2.35		1.0		1.0		1.0		1.0	

Cereal	5.0		4.5		5.5		5.5		5.5		5.5	
Meat	4.5		4.0		4.8		4.8		4.5		5.0	
Agent Type Thresholds	**Give Up**	**Give In**	**Give Up**	**Give In**	**Give Up**	**Give In**	**Give Up**	**Give In**	**Give Up**	**Give In**	**Give Up**	**Give In**
Productionist Pine	0.30	0.10	0.30	0.10	0.28	0.12	0.28	0.12	0.28	0.12	0.30	0.10
Productionist Spr	0.30	0.10	0.30	0.10	0.28	0.12	0.28	0.12	0.28	0.12	0.30	0.10
Productionist Pin-Spr	0.30	0.10	0.30	0.10	0.28	0.12	0.28	0.12	0.28	0.12	0.30	0.10
Productionist Bor.Br.	0.26	0.10	0.26	0.10	0.24	0.12	0.24	0.12	0.24	0.12	0.26	0.10
Multi-objective Pin-Spr	0.29	0.10	0.29	0.10	0.27	0.12	0.27	0.12	0.27	0.12	0.29	0.10
Multi-objective Pin-Bor.Br	0.25	0.10	0.25	0.10	0.23	0.12	0.23	0.12	0.23	0.12	0.25	0.10
Multi-objective Spr-Bor.Br	0.25	0.10	0.25	0.10	0.23	0.12	0.23	0.12	0.23	0.12	0.25	0.10
Multi-objective Bor.Br.	0.25	0.10	0.25	0.10	0.23	0.12	0.23	0.12	0.23	0.12	0.25	0.10
Recreationalist Pin-Spr	0.24	0.30	0.24	0.30	0.22	0.32	0.22	0.32	0.22	0.32	0.24	0.30
Recreationalist Bor.Br.	0.20	0.30	0.20	0.30	0.18	0.32	0.18	0.32	0.18	0.32	0.20	0.30
Recreationalist Nem.Br.	0.20	0.30	0.20	0.30	0.18	0.32	0.18	0.32	0.18	0.32	0.20	0.30
Conservationist Bor.Br.	0.23	0.30	0.23	0.30	0.21	0.32	0.21	0.32	0.21	0.32	0.23	0.30
Conservationist Nem.Br.	0.23	0.30	0.23	0.30	0.21	0.32	0.21	0.32	0.21	0.32	0.23	0.30
Passive Pin-Bor.Br.	0.10	0.30	0.10	0.30	0.08	0.32	0.08	0.32	0.08	0.32	0.10	0.30
Passive Spr-Bor.Br.	0.10	0.30	0.10	0.30	0.08	0.32	0.08	0.32	0.08	0.32	0.10	0.30
Passive Bor. Br.	0.10	0.30	0.10	0.30	0.08	0.32	0.08	0.32	0.08	0.32	0.10	0.30
Passive Nem. Br.	0.10	0.30	0.10	0.30	0.08	0.32	0.08	0.32	0.08	0.32	0.10	0.30
Commercial Cereal	0.24	0.70	0.24	0.70	0.22	0.72	0.22	0.72	0.22	0.72	0.24	0.70
Non-commercial Cereal	0.00	0.90	0.00	0.90	0.00	0.92	0.00	0.92	0.00	0.92	0.00	0.90
Commercial Livestock	0.24	0.70	0.24	0.70	0.22	0.72	0.22	0.72	0.22	0.72	0.24	0.70
Non-Commercial Livestock	0.00	0.90	0.00	0.90	0.00	0.92	0.00	0.92	0.00	0.92	0.00	0.90
Farm can take over forest	NO		NO		YES		YES		NO		YES	

3 Results

3.1 Ecosystem service provision

Timber provision grows steadily for all forest types except nemoral broadleaf during the first third of the simulation period across scenarios, as standing timber stocks are harvested (Fig. 8). Harvests fall thereafter and grow again during the last third of the century for the reference scenario, while for the SSP-RCP scenarios they remain largely unchanged during this period. Nemoral broadleaf timber provision, however, grows only modestly for all scenarios throughout the simulation period, while remaining far below demand levels. Carbon sequestration for the reference scenario mainly decreases during the first half of the century and increases during the second half. For all other scenarios it decreases throughout, though more slowly during the second half of the century.

Biodiversity and recreation provision generally increase for approximately the first 20 years, decrease until the middle of the century and increase thereafter. While scenarios with greater challenges to climate change mitigation (i.e. SSPs 3 and 5) tend to cluster together, others show higher supply levels and more differentiated trajectories. Biodiversity demands are met under four scenarios including the reference, while demand for recreation is only met under RCP3-SSP8.5.

Cereal provision grows and in some cases starts to stabilise in the second half of the simulation, although under no scenario does it reach demand. Meat supply does however meet demand throughout the simulation.

3.2 Land-use change

Overall, major land-use change was largely concentrated in the northernmost region (i.e. Upper Norrland), while the southernmost (i.e. Götaland) experienced the smallest changes (Fig. 9a). Changes in SSP3-RCP4.5, SSP3-RCP8.5 and SSP5-RCP8.5 are largely similar, although uncertainty in the magnitude of change is larger under RCP8.5 in the southern half of the country due to more divergent underlying climate change projections.

Nationally, an increase in agriculture and a decrease in unmanaged land are found, with the most extensive use of previously unmanaged land in the reference and SSP1-RCP4.5

scenarios. For other scenarios, extensive conversion of unmanaged land concentrates in Norrland. Agricultural expansion is generally larger under scenarios with higher mitigation challenges, and tends to concentrate throughout scenarios in the middle of the country (i.e. Lower Norrland and Svealand).

Monocultural conifer plantations tend to show minimal or no change. Pine-spruce forests generally expand extensively in the north and decrease in the south. Under the reference scenario however, they expand throughout the country. Mixed conifer-broadleaf forests expand in Upper Norrland and Svealand, but have only small increases in Lower Norrland. Trends in change for these forests differ in Götaland, where expansion happens under the reference scenario and SSP1-RCP4.5, but no change is observed under SSP4-RCP4.5, and loss occurs under all other scenarios. Boreal broadleaf forests have no observable change. Nemoral broadleaf forests undergo either no change or expansion, the latter being most prominent under SSP1-RCP4.5.

Hotspots of land-use expansion in pine-spruce forest, mixed conifer-broadleaf forest, nemoral broadleaf forest, agriculture and unmanaged land are observable for the different scenarios (Fig. 10a). Western Norrland is a hotspot for pine-spruce forestry expansion across all scenarios. Pine-spruce forest hotspots occur also in the south under the reference scenario, and to a lesser extent under SSP4-RCP4.5, SSP3-RCP4.5, SSP3-RCP8.5 and SSP5-RCP8.5. Only one hotspot occurs of mixed conifer-broadleaf forest expansion in the south under SSP1-RCP4.5. This scenario also has several hotspots of nemoral broadleaf forests in Upper Norrland, Svealand and Götaland. Agricultural expansion hotspots are present in the southern regions under SSP1-RCP4.5, SSP3-RCP4.5 and SSP3-RCP8.5, while under SSP5-RCP8.5 they also occur towards the north. Finally, a hotspot of land abandonment exists consistently in south-eastern Götaland under all scenarios except the reference.

3.3 Changes in land owner functional roles

Upper Norrland was the only region where no land owner functional role had a discernible decrease across scenarios (Fig. 9b). In all other regions the percentage of productionists decreased, except in Götaland under the reference scenario. Productionist loss was strongest in Svealand and weakest in Götaland, reflecting a decrease in the proportion of productionists nationally under all scenarios except the reference. Multi-objective forest owners were most successful, substantially increasing in numbers except in the south under scenarios with greater challenges to climate change mitigation. This type experienced its largest increases in Upper Norrland.

Recreationalists decreased in the southern regions and increased in Upper Norrland, but their total numbers decreased under all scenarios except SSP1-RCP4.5 and the reference scenario. Conservationists experienced small or no changes in all regions and under all scenarios except for SSP1-RCP4.5, under which their numbers increased substantially, especially in Upper Norrland. The percentage of passive owners remained nearly unchanged under all SSP-RCP scenarios, but increased slightly under the reference scenario, mostly in Götaland.

Commercial farmers generally increased in numbers under all SSP-RCP scenarios, but decreased in southern regions under the reference scenario. Non-commercial farmers show very similar changes under all scenarios, increasing in the north, barely changing in the mid latitude regions, and decreasing in the south.

Hotspots of increase in productionist, multi-objective, passive, commercial farmer, and non-commercial farmer functional roles can be observed among all scenarios (Fig. 10b). The reference scenario resulted in hotspots of productionists in Upper Norrland, multi-objective owners in all regions, and passive owners in Lower Norrland and Götaland. Under SSP1-RCP4.5 and SSP4-RCP4.5 multi-objective owner hotspots occurred in all regions although to a much lesser extent in the southern ones, where commercial farmer hotspots also occur under SSP1-RCP4.5. SSP3-RCP4.5 had incremental hotspots in multi-objective owners in southern regions, and commercial farmers in all regions. SSP3-RCP8.5 differs from SSP3-RCP4.5 in that it has a hotspot of multi-objective owners in the south, and it has no commercial farmer hotspots in Upper Norrland. Finally, SSP5-RCP8.5 has hotspots of multi-objective owners in all regions, primarily northern ones, commercial farmers throughout the country, and non-commercial farmers in Upper Norrland.

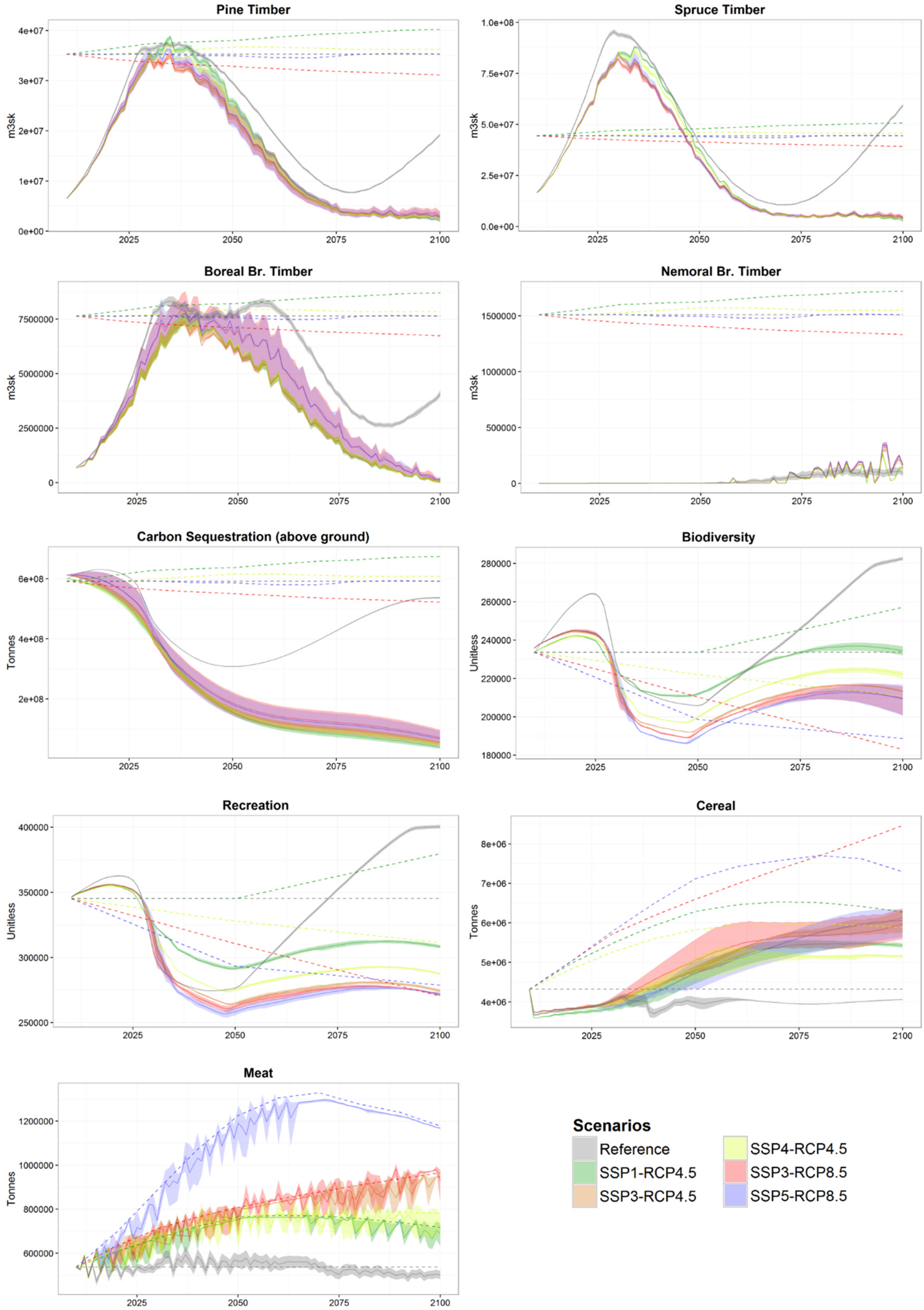
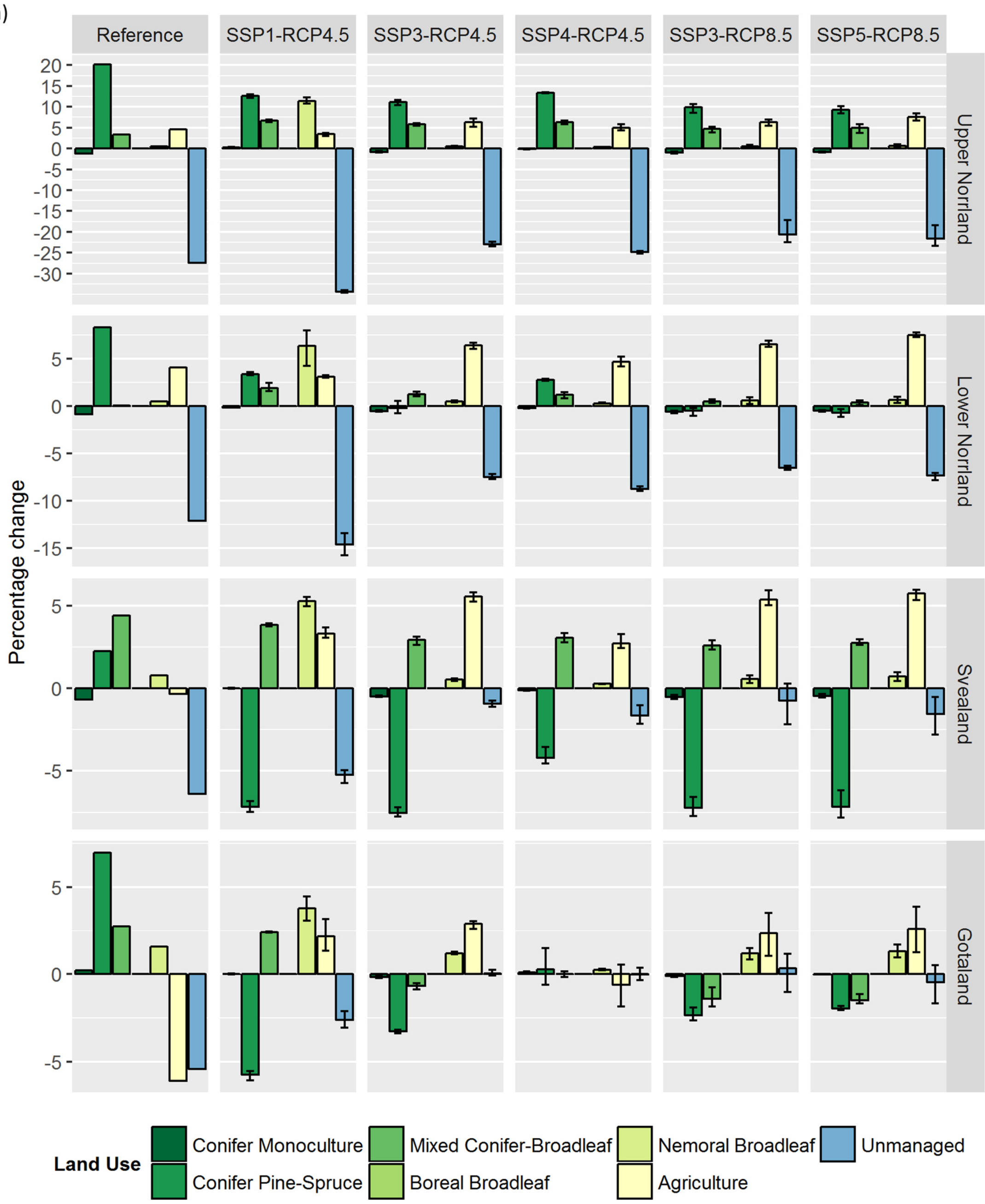


Fig. 8 Means (continuous lines) and ranges (semi-transparent areas) (defined by the three climate models) of ecosystem service provision, and service demands (dashed lines) through the study period, under the Reference and SSP-RCP scenarios

a)



Reference
SSP1-RCP4.5
SSP3-RCP4.5
SSP4-RCP4.5
SSP3-RCP8.5
SSP5-RCP8.5
Upper Norrland
Lower Norrland
Svealand
Gotaland
Percentage change
20
15
10
5
0
-5
-10
-15
-20
-25
-30
Land Use
Conifer Monoculture
Conifer Pine-Spruce
Mixed Conifer-Broadleaf
Boreal Broadleaf
Nemoral Broadleaf
Agriculture
Unmanaged

b)

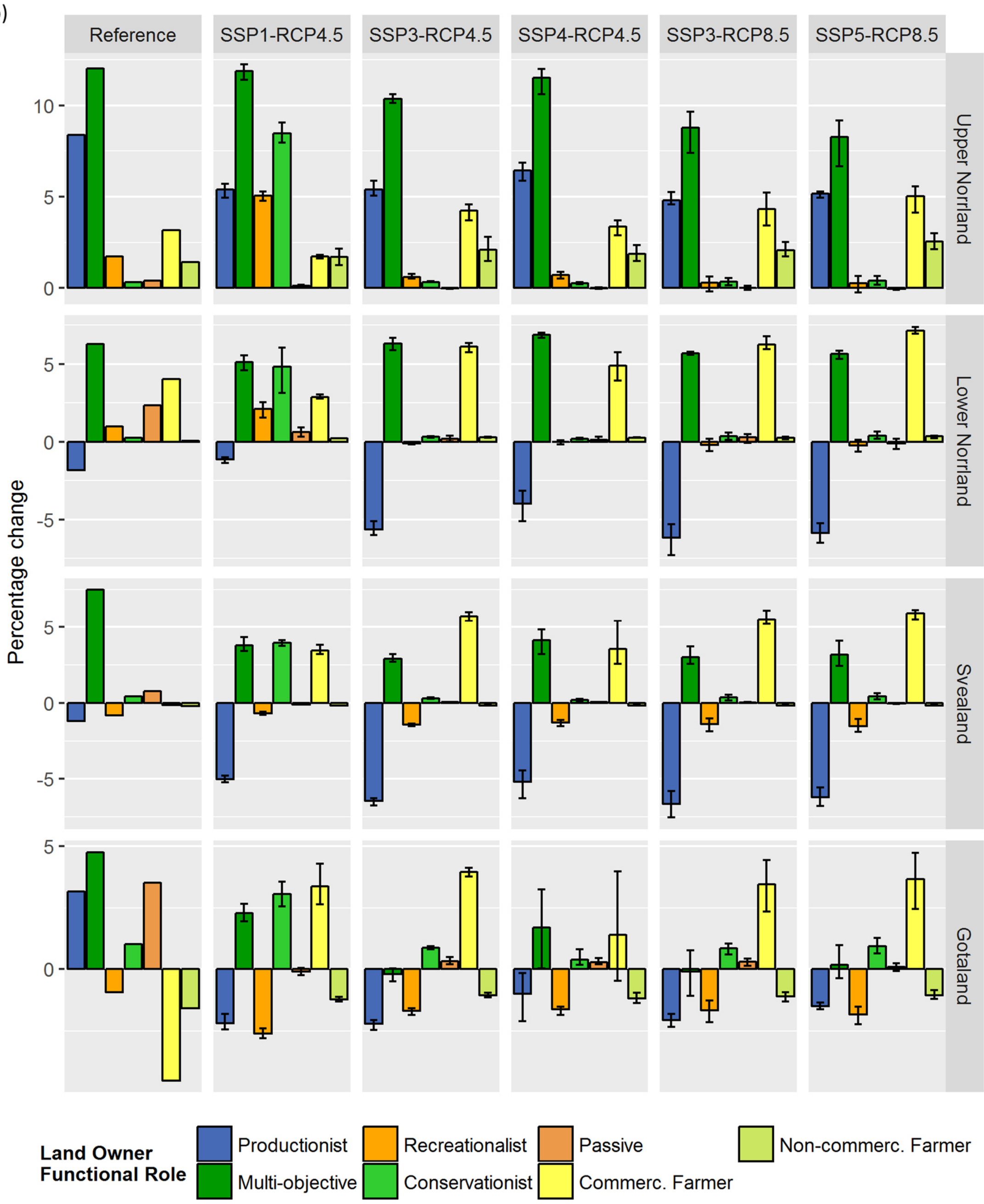
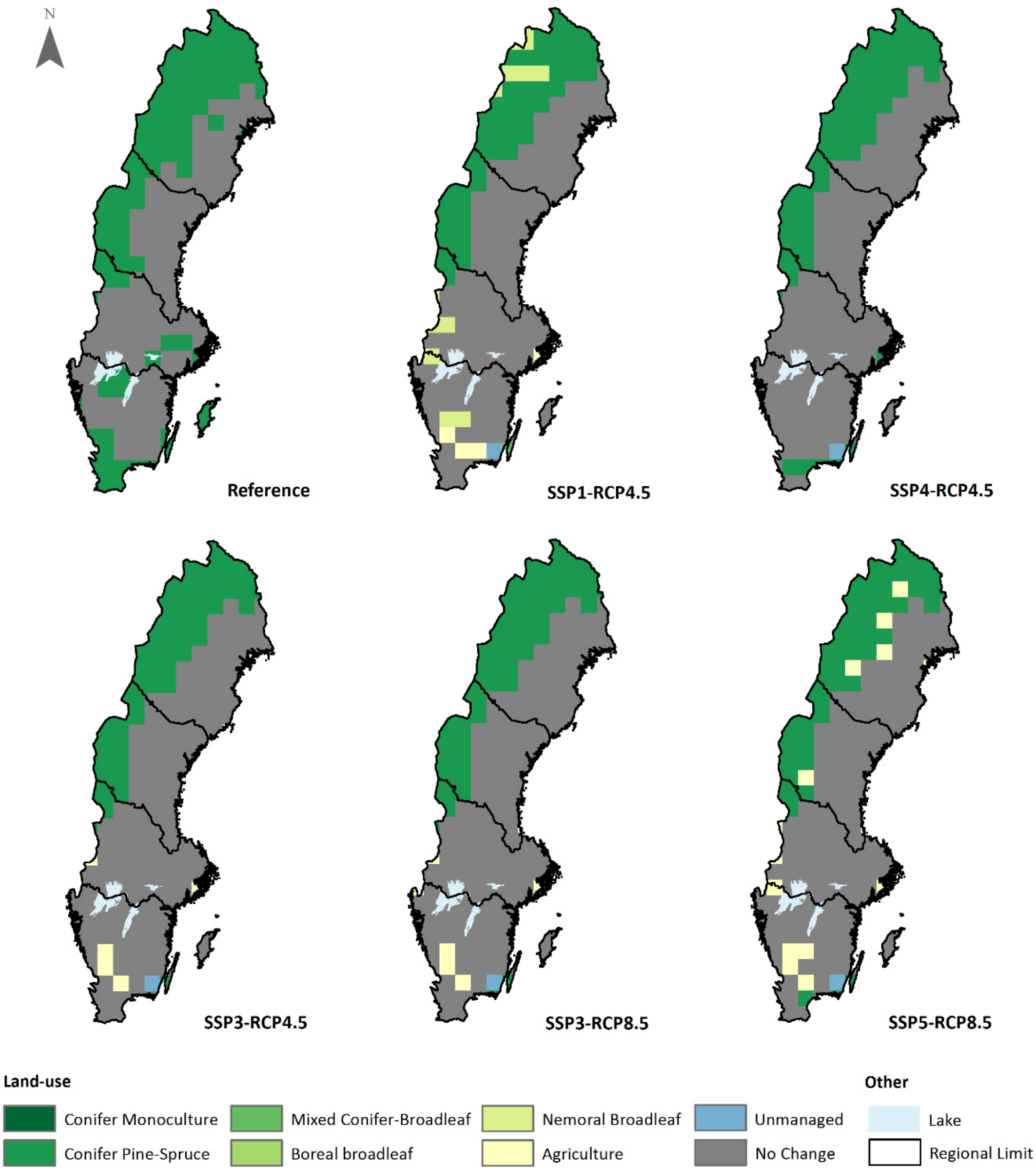


Fig. 9 Percentage change of a) land-use and b) land owner functional role categories per scenario and region. Error bars show the ranges of change generated accross the three climate models for each scenario

a)


N
Reference
SSP1-RCP4.5
SSP4-RCP4.5
SSP3-RCP4.5
SSP3-RCP8.5
SSP5-RCP8.5
Land-use
Conifer Monoculture
Conifer Pine-Spruce
Mixed Conifer-Broadleaf
Boreal broadleaf
Nemoral Broadleaf
Agriculture
Unmanaged
No Change
Other
Lake
Regional Limit

b)

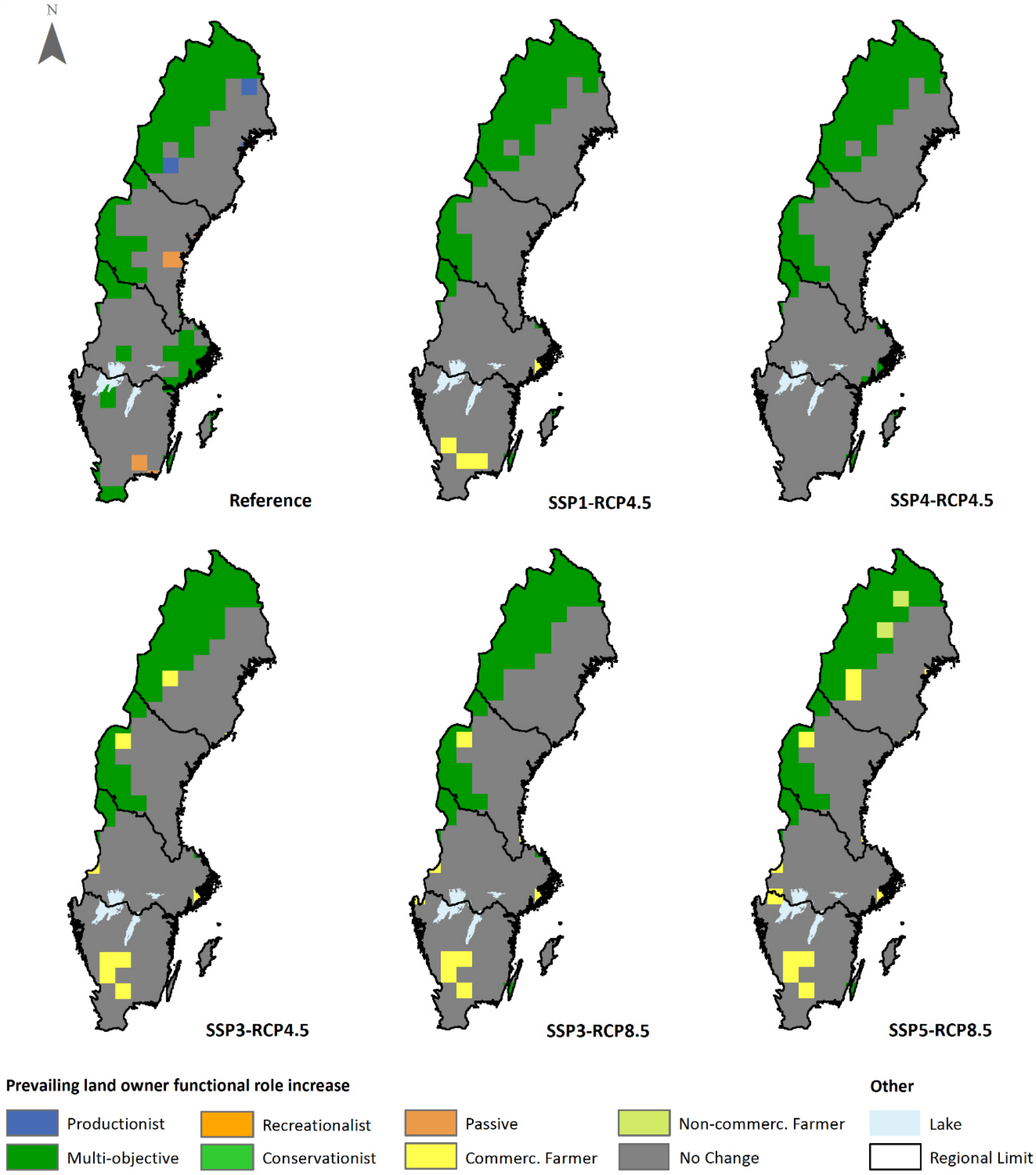


Fig. 10 Locations of hotspots of increase in a) land-use and b) land owner functional role categories under all scenarios in Sweden

4 Discussion

4.1 Future changes in land use and ecosystem service provision

This work demonstrates the adoption of mechanisms to simulate forest owner decision-making in large scale land-use change and ES provision modelling, and uses these to assess possible changes in the Swedish forestry sector under different climate and socio-economic change scenarios. Findings suggest substantial scope for model applications of this kind, and also for great variation in ES provision in the Swedish forestry sector, and important land-use change throughout the country.

Some of the changes in ES provision that I simulate are not caused by scenario conditions or forest manager decision-making. In particular, the sinusoidal trajectory of pine, spruce and boreal broadleaf timber supply through time under the reference scenario is largely determined by the uneven distribution of forest ages within Sweden. This is a legacy of past land management decisions. Forestation in the country increased from the beginning of the 20th century until the 1960s, when the maximum area of forest planted per year was attained, after which rates began to decline (Swedish Forest Agency 2015). The model suggests that this will result in a peak in supply during the 2030s, approximately 70 years after the forest planting peak. In reality, the magnitude of this peak is likely to depend upon the extent to which felling can be coordinated to preserve supply and price levels, but the need to harvest within certain time periods does constrain the scope for such actions. As a result, decreases are likely in timber revenue and in the supply of other forest services, with reductions in the supply of biodiversity, recreation, and carbon sequestration being projected in our simulations. Therefore, it is clearly important that the impacts of a future acceleration in timber harvesting are considered and addressed.

More generally, the substantial decoupling between the simulated supply of, and demand for, forest services over the course of 90 years (in contrast to the results for agriculture) demonstrates the difficulty of continuously meeting societal demands for ES produced over long periods of time. The consistent supply of multiple forest services represents a complex optimisation problem that can only be solved, if at all, using national overviews and top-down (e.g. policy) mechanisms. An obvious example is the provision of biodiversity and recreation, which depends upon nationally designated areas and prohibitions on felling in non-productive forests (Swedish Forest Agency 2014c), even in the most environmentally-friendly scenarios I modelled (i.e. SSP1-RCP4.5). Future analyses of service provision under such scenarios should therefore also consider services provided by protected natural systems, in order to evaluate how far they are able to absorb impacts of changes such as large-scale felling in productive forests. Another consideration is the extent to which food demand exerts pressure on forest management and ES provision, with our simulations suggesting shortfalls in cereal supply even under sub-optimal forest ES provision. A higher societal sensitivity to cereal supply levels would contribute to bringing supply closer to the demand, which would further compromise the provision of forest ES. Furthermore, the potential increase in agriculture in the north would entail a more widespread competition with forestry throughout the country. Our simulations suggest that this northward expansion would largely come at the expense of unmanaged land, comprising wetlands and semi-natural vegetation, which also supply ES such as water supply, nutrient retention, food, recreation, or biodiversity (Costanza et al. 1997). Hence, it is important to understand that meeting future demands for agricultural and forest services will likely entail trade-offs with ES supplied by other natural systems.

I also find that, under the SSP-RCP scenarios, carbon sequestration decreases as timber is increasingly felled throughout the first third of the century, but does not increase again as timber felling decreases after that (in contrast to the reference scenario). Similarly, biodiversity and recreation decreases are never entirely reversed under the SSP-RCP scenarios, while they are under the reference scenario. The reason for this is that forest-to-forest land-use change is substantially more frequent under the SSP-RCP scenarios than for the reference, meaning that fewer forests reach their scheduled age of felling under the SSP-RCP scenarios. The lower mean national forest age reached under SSP-RCP scenarios also entailed smaller annual timber harvests nationally as younger forests were being felled, and this keeps carbon sequestration, biodiversity and recreation at lower levels. These findings confirm qualitative results from Roberge et al. (2016), who suggest that the provision of supporting and cultural ES, and economic outputs from timber would be negatively impacted by shorter rotations in Fennoscandian forests.

This phenomenon is explained by changes in demand levels and climate change, which together prompt forest managers to adapt their management activities more frequently and dramatically than they would otherwise do (with the size of this effect depending on the balance between the costs of early felling, timber prices and potential profits from alternative management strategies).This effect is consistent with the ongoing consideration of management alternatives (e.g. multi-species planting, or introduction of exotic species respectively) as adaptations to climate change (Felton et al. 2016; Kjaer et al. 2014). If forest owners do consider felling early when faced with a profitable alternative, the successful uptake of one or more innovations could trigger continued changes between forest types as simulated here, which could negatively affect the provision of services that increase with forest age. Further simulations including the uptake of innovations as a function of networks among forest owners (Satake et al. 2007) and relevant institutions would be particularly useful here.

Many of our findings were broadly consistent between SSP3-RCP4.5 and SSP3-RCP8.5, which seems to indicate that the influence of climatic change on land productivities (being the only parameter different between them) may be less important than that of socio-economic changes or behavioural differences. Additionally, the great resemblance between SSP3-RCP8.5 and SSP5-RCP8.5 suggests that land owner and societal behavioural factors (e.g. sensitivity to profit levels on the part of landowners and sensitivity to supply levels on the part of society), despite substantially different service demand trajectories, are key in determining the course and impact of land use change. Furthermore, this holds true for the provision of timber and non-timber forest services.

Climate change, and especially service demand levels do, however, appear to influence the geographical distribution of land-use, as observed when comparing SSP3-RCP4.5, SSP3-RCP8.5, and SSP5-RCP8.5. We see an expansion in both commercial and non-commercial agriculture towards the north as a response to the higher demands for agricultural products and increasing suitability for agriculture in the north. Additionally, the distinctive expansion of nemoral broadleaf forests and conservationists across the country, and of recreationalists in the north under SSP1-RCP4.5, are consequences of the higher value and demand being placed by society on biodiversity and recreation.

4.2 Model assumptions and limitations

It can be difficult to identify causes and effects in simulations from complex system models because of the multiple interactions and feedbacks inherent within such models. However, some relationships can be identified between model inputs and outputs from the sensitivity analysis and the exploration of simulation results, and sources of uncertainty among model assumptions. Giving-in and giving-up thresholds represent a range of personal characteristics that control an agent's responsiveness to demand levels. Here I assign random distributions to these thresholds, in the absence of empirical data with which they could be parameterised. Our, and previous (Brown et al. 2016c; Brown et al. 2014; Murray-Rust et al. 2014), sensitivity analyses show that the effect of random model components on agent behavioural parameters has a lower impact on ES provision compared with components that differ between scenarios (demand and productivity changes, and mean behavioural values). Land owner giving-up probabilities help to regulate the rate of owner type and land-use change, with lower giving-up probabilities resulting in less abrupt yearly changes, especially for farmers/farmland. Greater randomness between agents means less 'rationality' in land use change. Further insight into how behavioural parameters affect simulation results, through extensive sensitivity analyses, are presented in Brown et al. (2014), Brown et al. (2016c) and Murray-Rust et al. (2014).

The ES demand change scenarios were derived from the SSP-RCP scenarios, but it is important to acknowledge the difficulty in estimating such demands because of the substantial uncertainty inherent within these scenario assumptions. It is also important to be cautious with the results for owner type and land-use change in the north west of Sweden, dominated by the Scandes mountain range. In this area, the effect of topography on productivity changes arising from climate change is difficult to model since productivities were calculated from climate change data at a coarser spatial resolution (50x50 km). Therefore, the effect of climate change is highly uncertain in north-western Sweden. Furthermore, the model allocates forest in the mountainous region under all scenarios, including the reference, to cells where forest was previously not present. This is mainly due to the time lag between the moment when forests start being planted in response to unmet demands and the moment when demands are met. During this time period, while service supply from young forests is still low, forests continue to be planted, to the point that they occupy available areas that may be less productive. This may be unrealistic depending on the level of return that 'real world' land owners are willing to accept from forest activities.

Climate change is expected to have a substantial impact on biodiversity, disrupting the equilibrium of biomes through species extinctions, shifts in species and community distributions, phenological disturbances, or interspecific relationships among other (e.g.

Bellard et al. 2012; Dawson et al. 2011; Millennium Ecosystem Assessment 2005). Our model did reflect the effect of climatic change on the distribution of differently biodiverse forest types, leading to shifts in the location and aggregate amount of biodiversity being generated. Otherwise, we only used biodiversity indicators that were representative for the scale and resolution of our approach, i.e. coarse woody debris levels, tree diversity, management practices, and (having a much smaller effect) forest productivity. While the generation of coarse woody debris (a determinant of biodiversity in our ABM) may be affected to some extent by climate change as timber volume growth rates increase, its availability will still be mainly determined by the management strategy being applied (Mazziotta et al. 2015). Tree diversity and management practices are indirectly affected in the ABM through the limitations posed by forest productivity. The direct effect of forest productivity on biodiversity was very small, and therefore barely if at all representative for the direct effect off climate change on biodiversity. Overall, given the scale used and the subsequent limited small-scale biological realism, it was difficult to accurately represent the effects of climate change on biodiversity beyond its effect on forest distribution. Therefore, if the levels of biodiversity generated by particular forest types were altered by climate change, resulting aggregate levels of biodiversity would be more uncertain. If this were the case, the competitiveness of different owner types could differ too through time, leading to different land-use transitions and levels of service production.

Not allowing land ownership to change between forest owner types before stand maturation means that the effect of management on forest growth rates and their service provision throughout the life of the forest is fixed when the forest is planted. This would only be a limitation though if it were common for forest ownership and management to change during early forest development stages. Additionally, I did not account for voluntary set-asides for conservation, which make up at least 5% of the productive forest area (Swedish Environmental Protection Agency 2016). In addition, due to certification requirements, trees, tree groups and buffer zones are left on most felling sites, also reducing the harvested volume somewhat. Had voluntary set-asides and certification requirements been accounted for, yearly timber provision would have been lower and the outcomes of the modelled competition process slightly different. Conversely, as today, the majority of the planting material in Sweden comes from seed orchards, and these plants are expected to grow considerably faster (10-20%) compared to plants used in past decades, accounting for this effect would have led to higher standing volumes and timber supply. Consequently, the effect on timber provision of not including voluntary set-asides or certification requirements, and the effect of not including planting material from seed orchards have opposing effects at the aggregate level.

5 Conclusions

CRAFTY-Sweden brings us a step forward in the understanding and representation of large scale land-use change and its complexities under climate change. Our results show that variability in human behaviour has a substantial role in determining the effects of climatic forces and societal demands on ES provision and land-use change. Important changes in land-use and ES provision can be expected in Sweden as a result of changing demands and climate, especially towards the north of the county. Increasing food demands and increasingly favourable climatic conditions for agriculture would lead to agricultural expansion, if food imports are not increased. Such expansion would require trade-offs with currently unmanaged land, such as wetlands, which contribute important ES. Furthermore, accelerating timber harvesting throughout Sweden due to a nationally uneven age distribution, and increasing rates of forest land-use change between forest types may have negative consequences for forest ES provision. Finally, the challenge of steadily meeting societal demands for ES produced over long time periods and at large scales would require top-down mechanisms that use national-scale information to regulate forestry processes and, to the extent possible, their consequences.

Chapter 4

The importance of socio-ecological system dynamics in understanding coping and adaptation to global change in the forestry sector

---

This chapter has been submitted for publication to the *Journal of Environmental Management*. It builds on the methodology presented on Chapter 3 to present and discuss new modelling results with a greater focus on adaptation to global change.

1 Introduction

Adaptation is needed to offset or exploit the effects of climate change on socio-ecological systems. This is especially so in forestry, a sector that is sensitive to the ecological and economic impacts of climate change (Hanewinkel et al. 2013; Keskitalo 2011; Lindner et al. 2010), and where timespans of several decades exist between planting decisions and harvesting. Forestry also has great potential for climate change mitigation, through carbon sequestration and the use of wood biomass as a renewable energy source. However, competition for land with agriculture is likely to intensify as food demands grow, further altering the distribution and composition of forests and the levels of ecosystem services (ES) they can provide (Alexandratos and Bruinsma 2012; Buonocore et al. 2012; Tilman et al. 2001; Zanchi et al. 2012). However, despite the importance and sensitivity of managed forests to global change, relatively little is known about how the sector can or will adapt. It is important therefore to improve understanding of forestry adaptation processes, the drivers and consequences of forest owner decisions, and the suitability of different forest management strategies in meeting societal service demands under future climatic and socio-economic conditions.

Effective management strategies enhance the coping capacity of managers and the land system. Knowledge about effective management strategies can also contribute to building resilience through adaptive capacity in land-based sectors. The concepts of coping and adaptive capacity have different meanings and connotations, and yet they have often been used to mean the same thing (Levina and Tirpak 2006). Coping capacity can be defined as "the ability of people, organizations, and systems, using available skills, resources, and opportunities, to address, manage, and overcome adverse conditions" (IPCC 2012), p. 558). Adaptive capacity can be understood as "the combination of the strengths, attributes, and resources available to an individual, community, society, or organization that can be used to prepare for and undertake actions to reduce adverse impacts, moderate harm, or exploit beneficial opportunities" (IPCC 2012), p. 556). Coping capacity can be strengthened with adaptation measures, while adaptive capacity includes both coping capacity and potential adaptation measures (Levina and Tirpak 2006). As a result, both are dynamic processes, with adaptive capacity in particular evolving as a result of changes in climate impact, society, economy, policies, etc. (van Gameren and Zaccai 2015).

The concept of adaptive capacity has been used to evaluate generic, sectoral and cross-sectoral adaptation (e.g. IPCC 2012; Johnston and Hesseln 2012; Lindner et al. 2010; Sharma and Patwardhan 2008). However, the estimation of these capacities is commonly done through indicators, aggregated into indices, which are specific to the (temporal) context in which they are defined (Acosta-Michlik et al. 2013; Metzger et al. 2006; Vincent 2007). While the assessment of past and present adaptive capacity through such indices is accepted, no appropriate method currently exists for their projection into the future, which requires consideration of complex system dynamics affecting indicator variables (Araya-Muñoz et al. 2016; Vincent 2007). Furthermore, while some studies have considered present capacities in combination with impact projections to assess future vulnerability (Lung et al. 2013; Metzger et al. 2006; Preston et al. 2008), the (unquantified) uncertainty carried by these approaches limit their utility (Lung et al. 2013). Besides, the implementation of adaptive measures will depend on a range of uncertain behavioural and cognitive factors (e.g. beliefs, experiences) (Blennow 2012; Vincent 2007), which are not considered in quantitative assessments of adaptive capacity. Hence, the concept of capacity as commonly employed is not sufficient for studies of future adaptation processes.

Instead, future adaptation processes may be explored through process-based models of socio-ecological systems, which allow simulation not only of the states of the system, but also the dynamics that determine state changes. In land-based sectors, adaptation (including maladaptation) involves changes in land cover and/or management strategies (i.e. land-use change) that are determined by the decisions of land managers. In forestry, the capacity to cope or adapt to future environmental conditions under different management strategies has been evaluated (Le Goff et al. 2005; Seidl and Lexer 2013; Valladares 2008), but the behaviour and objectives that constitute the decision-making processes of forest owners about such management strategies (Andersson and Gong 2010; Ingemarson et al. 2006; Vulturius et al. in review) have seldom been considered in such assessments. Only Rammer and Seidl (2015) simulated adaptive management to climate change in forest landscapes by coupling human and environmental systems using agent-based models (ABM; i.e. models in which the individual behaviour of agents such as land owners is represented explicitly). In no cases have models been used to assess the suitability of particular management strategies for climate change adaptation. Furthermore, despite the importance of regional and global drivers of forest change such as competition for land or societal demands for ES, only climate change alone has been assessed in models of adaptive forestry systems.

I present therefore a novel approach to adaptation assessment based on a land-use ABM to assess the suitability of different forest management strategies in adapting to future socio-economic and climatic change. The model accounts for land owner behaviour, the supply of, and demands for, ES, and the effects of climate change on land productivity. I apply this model to the Swedish forestry sector and its competition with agriculture. Simulations were undertaken for different socio-economic and climatic scenarios, to evaluate the competitiveness of different forest management strategies and their evolution through time. The model essentially represents autonomous adaptation processes, i.e. “adaptation that does not constitute a conscious response to climatic stimuli, but is triggered by ecological changes in natural systems and by market or welfare changes in human systems” (IPCC 2007)p. 869). Hence, an additional purpose of the work presented here is to assess how useful measures of competitiveness and coping ability are in illuminating adaptation processes in forestry, and autonomous adaptation in particular.

2 Methods

I used the CRAFTY-Sweden model (Chapter 3) to explore adaptation to global change in the Swedish forestry sector. CRAFTY-Sweden is an applied extension of the CRAFTY ABM framework (Murray-Rust et al. 2014) that represents large-scale land-use dynamics, based on the demand and supply of ES (Fig. 11). The services represented here are timber (from several different tree species), carbon sequestration, biodiversity, recreation, cereal and meat production (from agriculture). Demands are defined exogenously while supply depends on land productivity (that varies through time and space), the behaviour of modelled agents, and the infrastructure present at a given location.

Agents include different types of forest owners and farmers characterised by their objectives and associated management practices (Chapter 3), with farmer agents included in order to simulate the competition for land between forestry and agriculture. Geographical space is represented as a grid of cells across the whole of Sweden at a resolution of $1km^2$. Each cell has defined levels for a range of capitals representing the availability of infrastructure and land productivity. A cell is managed by a single land-use agent, which uses the capital stock available within the cell to provide services according to its own production function. The competitiveness of a given level of service provision can be calculated based on societal demands and overall supply levels through 'benefit' functions, which describe the monetary and non-monetary value to society of service production. Agents can make decisions based on their current competitiveness (including an estimation of the future benefits they expect to attain from their land-use), and their willingness to abandon, change management or hand over land to a different owner. On the basis of their competitiveness, agents participate in an allocation procedure with potential new agents that results in land-use change.

In the following, I outline the land owner typology from which modelled agents are drawn, and the production and behavioural mechanisms. I then describe the approach to scenarios and the analysis of simulation results. For further detail regarding CRAFTY-Sweden see Chapter 3.

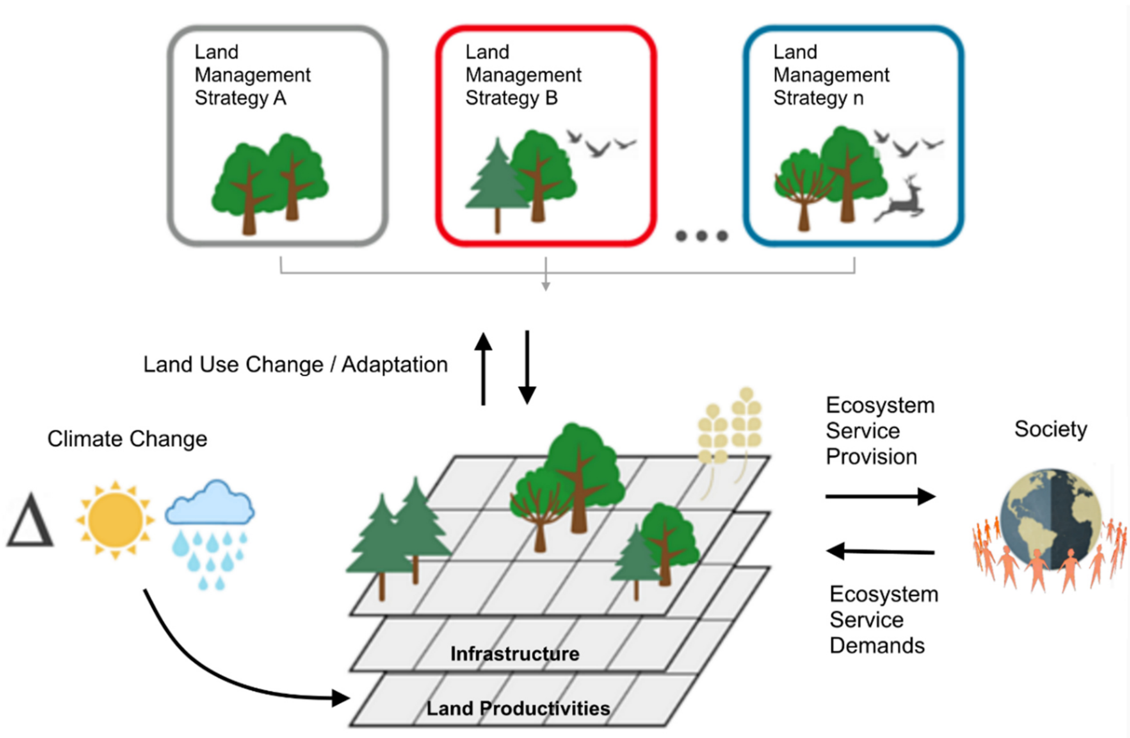


Fig. 11 Schematic representation of the drivers and their interactions in the CRAFTY-Sweden modelling approach

2.1 Land owner typology

Forest owners were classified into 17 forest owner types (Table 9), and farmers into four farmer types (i.e. commercial cereal, non-commercial cereal, commercial livestock, and non-commercial livestock). Details regarding the development of the land owner typology see

Chapters 2 and 3. The management and behavioural mechanisms of owner types are described in sections 2.2 and 2.3.

Table 9 Possible combinations of overarching forest management roles (columns) and forest types (rows) found to be managed under such roles in Sweden. Each combination constitutes a management strategy

	Productionist	Multi-objective	Recreationalist	Conservationist	Passive
Pine	■				
Spruce	■				
Pine-Spruce	■	■	■		
Pine-Boreal Broadleaf		■			■
Spruce-Boreal Broadleaf		■			■
Boreal Broadleaf	■	■	■	■	■
Nemoral Broadleaf			■	■	■

2.2 Land owner service production

The production of agricultural services was modelled on an annual basis. Forestry services are however dependent on forest age and hence are less consistent through time (e.g. the changing provision of recreation, and the highly irregular provision of timber when a forest is thinned or felled). Additionally, climatic change can affect service supply by acting on productivities.

The production of a service within a cell is dependent on the optimal production (i.e. production under ideal conditions) that an agent type would be able to achieve in a given year, affected by cell capitals, the annual climate-induced change in cell capitals, and the capital sensitivities of that agent type. To reflect individual variability, optimal production and capital sensitivity levels were randomly drawn from uniform distributions. The following section describes the main elements underpinning the calculation of the production of different ES. Further details are available in Chapter 3.

For timber production, optimal production is given by a forest owner type-specific function that determines timber growth given a forest's age. The ProdMod model (Eko 1985) was used to generate timber growth curves for each owner type given their management preferences (including different parameters relating to forest planting and thinning operations) (Table 7) under currently ideal environmental conditions. Optimal production functions of above-ground carbon storage were also calculated using ProdMod. The forest in a cell is clear-felled when it reaches an age that depends on site productivity and owner objectives. Given recommendations and legal regulations of stand age at felling to guarantee that the production potential is utilised, I defined (productivity dependent) minimum felling ages for the different forest types. Felling age is determined at the time that an agent is allocated to a cell by randomly drawing a number (i.e. age) from within an agent type-specific Gaussian distribution of the planned felling age (above minimum felling age). Felling age-related parameters are given in Table 7. Upon felling, timber is harvested and carbon stored in the standing timber is removed from the national pool.

The calculation of optimal forest biodiversity production considered forest age, using the generation of coarse woody debris with age, tree diversity and management practices undertaken by each owner type (e.g. woody debris removal), which have an influence on biodiversity. Recreational value was largely determined by forest age, but also by management practices, accessibility and, to a lesser degree, by the types of trees and combinations present (i.e. conifer vs broadleaf, and monoculture vs mixed).

According to baseline maps with available capitals and agricultural agent type locations, the optimal productions and capital sensitivities of agent types were adjusted so that total cereal and meat production equalled the total production in Sweden reported by the FAO (2015) for 2010. The production of non-commercial agents was set at 0.6 times that of the commercial agents to reflect approximate differences in production potentials across equivalent classes in Van Asselen and Verburg (2013).

2.3 Competition for land

Farm land can be taken over by other agents each year because agricultural management is on annual timescales. For forest owners, however, I assume no abandonment or change in management approach until the forest has reached maturity, in order for forest agents to recover their initial investment. At this point a 'potential' agent with higher competitiveness than the incumbent agent can take over the land, resulting in one of two outcomes:

1. If the potential agent is a forest owner type willing to plant the same forest type as that already standing in the cell, it will inherit the production functions of the former

owner, as the effect of changing management of a forest once maturity is reached is negligible. Age of felling is however adjusted to meet the objectives of the new agent. Because passive owners are assumed to acquire forests through inheritance, they follow this system exclusively and do not compete for unmanaged land.

2. If the potential agent is a farmer or a forester not meeting the above criteria, the standing forest is clear-felled and land is either converted to farmland or to newly-planted forest.

Forest owners plan what they will plant according to (non-climate sensitive) data that show potential tree growth according to site conditions. While some owners may also consider climate change and risk spreading, this is currently not a generalizable trait of Swedish forest owner decision-making (Blennow et al. 2012). Hence, while farmer service production is evaluated for the coming year, forest owners evaluate it for the (future) year of felling under current conditions. To evaluate agent competitiveness for a given bundle of services a benefit function is used, with production level being time-discounted by forest age at planned felling to reflect the desire for shorter-term returns where possible. Time-discounted production is normalised by the current per-cell demand for particular services to achieve levels that are comparable across agent types supplying services that are measured in different units. Accordingly, a per-cell unmet demand is taken into consideration, and normalised by total demand to give a proportional unmet demand. A weighting factor representing the assumed importance to society of meeting service demand levels of each service is also part of the benefit functions. For further details regarding benefit functions and parameter values used see Chapter 3.

If an existing agent's competitiveness is lower than its 'giving-up' threshold, it will abandon the cell. If a potential agent's competitiveness within a cell is greater than the existing agent's by a value larger than the existing agent's 'giving-in' threshold, then the potential agent takes over the cell (subject to the time constraints described above). Giving-up and giving-in thresholds reflect minimum acceptable benefit and tolerance to competition respectively, and are drawn from agent type-specific Gaussian probability distributions to simulate individual differences (Murray-Rust et al. 2014). Also, since farmers and foresters are not all affected by market conditions to the same degree or at the same time, I implement giving-up probabilities for each agent type that apply to agents whose competitiveness falls below their giving-up threshold. Mean giving-in and giving-up thresholds are given in Table 8, and giving-up probabilities are given in Table 7.

2.4 Scenario analysis

Five future scenarios were defined by combining Representative Concentration Pathways (RCPs) and Shared Socio-economic Pathways (SSPs) as described in chapter 3. Each RCP was simulated with three climate models to capture some of the uncertainty across climate models. Each climate model-RCP combination consisted of a different set of climate-induced annual productivity changes.

The ecosystem model LPJ-GUESS (Smith et al. 2001) was used to simulate forest dynamics during 2010-2100 using climate projections from the Global Circulation Model - Regional Circulation Model ensembles (hereupon 'climate models') EC-Earth-RCA4, IPSL-RCA4 and NorESM-RCA4 for RCPs 4.5 and 8.5 from the EURO-CORDEX project (Jacob et al. 2014; Jones et al. 2011). Following land-use and European SSP storylines from Engström et al. (2016) and Kok et al. (2015) respectively, SSPs differed in: a) future demands for ES, b) the importance to 'society' of meeting demands for each service, c) probability distributions for owner type giving-in and giving-up thresholds, and d) the potential for farmland to displace forest land (Table 10).

Table 10 SSP scenario descriptions in relation to ecosystem service demands, importance to society of meeting service demand levels, and giving-up and giving-in thresholds (reflecting willingness to sell land and tolerance to competition respectively). For parameter values see Table 8

Reference	Demands for all services remain unchanged through time. Higher importance of meeting demands for pine and spruce timber than for boreal and nemoral broadleaf timber. Medium importance for all other services. More profit-oriented owner types, being more sensitive to benefit values, have higher giving-up thresholds and lower giving-in thresholds. Farmland cannot displace forestry.
SSP1 "Sustainability"	Demands for timber and carbon sequestration grow until the end of the century. Demands for biodiversity and recreation are stable until 2050 and grow thereafter. Cereal and meat demands grow until the middle of the century and slowly decrease afterwards. Farmland cannot displace forestry. Compared to the Reference, higher importance for carbon sequestration, biodiversity and recreation, and lower for cereal and meat. Similar giving-up and giving-in thresholds as under the Reference.
SSP3	Demands for timber, carbon sequestration, biodiversity and recreation decrease, and cereal and meat demands grow, until the end of the century.

"Regional Rivalry"	Compared to the Reference, lower importance of meeting demands for carbon sequestration, biodiversity and recreation, and higher for cereal and meat. Lower giving-up and higher giving-in thresholds than under the Reference Farmland can displace forestry.
SSP4 "Inequality"	Demands for timber and carbon sequestration grow (less than under SSP1-RCP4.5) until 2050, and decline thereafter. Demands for biodiversity and recreation decrease throughout. Demands for cereal and meat grow until the middle of the century (less than under SSP1-RCP4.5 for cereal, but equally for meat) and stabilise thereafter. Compared to the Reference, lower importance of meeting demands for timber, carbon sequestration, biodiversity and recreation, and higher for cereal. Similar giving-up and giving-in thresholds as under SSP3-RCP4.5. Farmland cannot displace forestry.
SSP5 "Fossil-fuelled development"	Demands for timber, carbon sequestration, decline during the first half of the century and grow back to initial levels afterwards. Biodiversity and recreational demands decrease over the century, but at a higher rate during the first half. Demand for cereal grows substantially during the first half of the century and remains relatively stable thereafter. Meat demands more than doubles in the course of the first sixty years and slowly declines thereafter. Similar to SSP3-RCP4.5, but higher importance of meeting demands for meat. Similar giving-up and giving-in thresholds as under the Reference. Farmland can displace forestry.

CRAFTY-Sweden simulations were run for the 2010-2100 period at a 1km^2 resolution. The model was calibrated to produce minimal short-term (decadal) changes in land management under constant levels of demand and productivities, so that the effects of long-term forest management and scenario conditions could be isolated. The model was then run under these static conditions for the period 2010-2100 to produce a reference scenario. To measure the effect of stochastic model components 32 simulations of the reference scenario were run under different random seeds, but otherwise identical parameterisations. Finally, each climate model-RCP-SSP combination was run once (under one random seed).

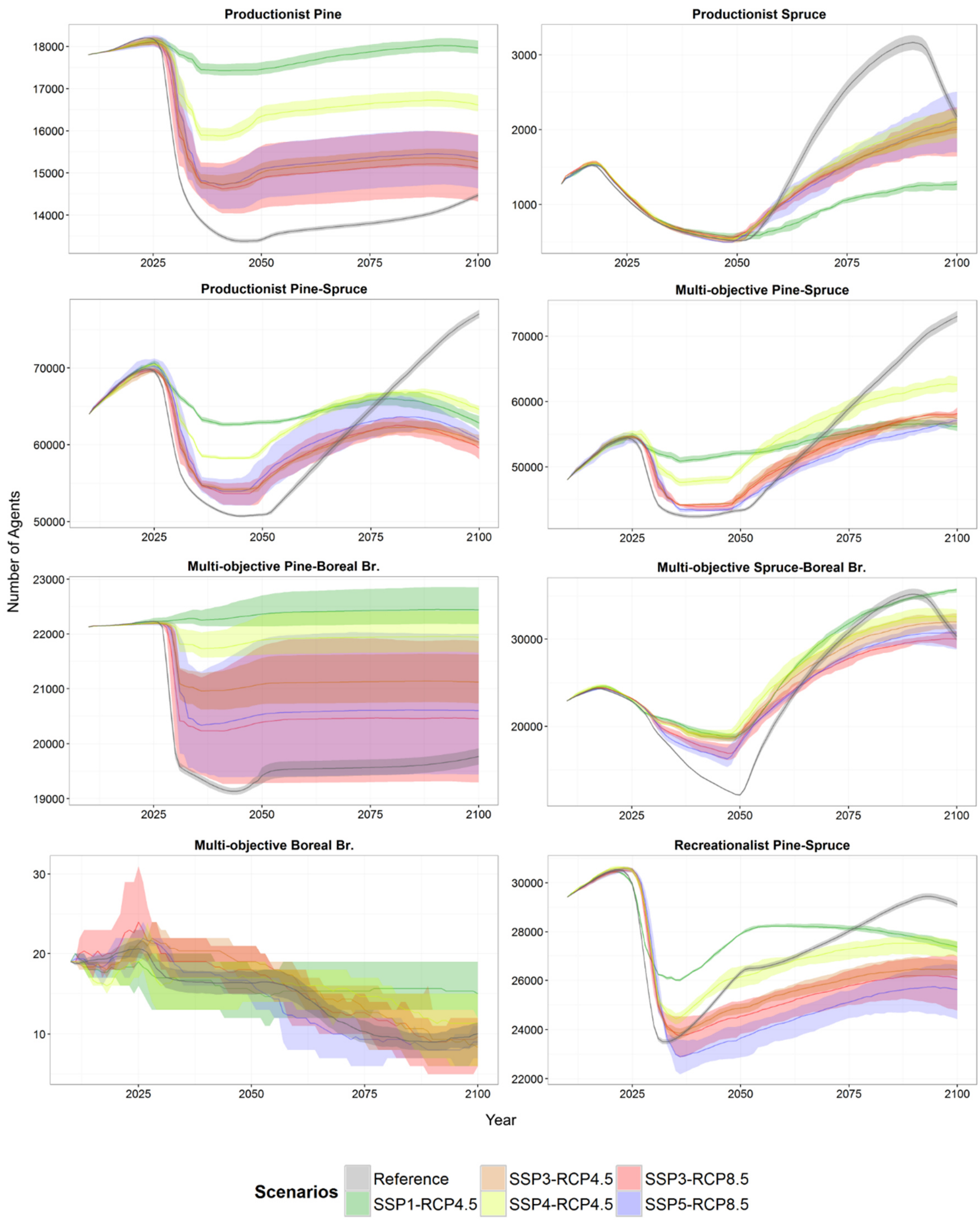
2.5 CRAFTY-Sweden outputs

Modelling outputs presented here relate to forest owner competitiveness and *coping ability*, which together determine the course of autonomous adaptation. To evaluate competitiveness, agent types were mapped for every year during 2010-2100, and are presented as ranges of owner type numbers defined by maximum and minimum values among climate models plotted for each scenario. Coping ability is assessed using a coping index, which reflects whether a particular management strategy is at least as competitive under an uncertain future global change scenario (defined by the scenario space) as under present conditions (defined by the reference scenario). Coping index levels through time were calculated by assigning a yearly score to each management strategy depending on the number of scenario simulations (out of 15), which return a number of owners implementing that strategy above the reference. A point was scored for every simulation where numbers were larger than or equal to those under the reference, and zero if they were smaller. The coping index was also assessed at the level of forest owner functional roles by aggregating index levels of management strategies belonging to each role weighted by the maximum number of agents achieved under a simulation at each point in time.

3 Results

3.1 Competitiveness

In general, I see substantial changes throughout the simulations with distinct trajectories in the numbers of owners implementing the different management practices (Fig. 12). All management strategies had different levels of competitiveness under different sets of objectives, given the same forest type. Multi-objective owners were always the most competitive managers for a particular forest type, while passive owners were always the least competitive. Other owners differed in relative competitiveness depending on their objectives and forest type. Owners managing forests containing pine or spruce showed an initial increase or no change in numbers until around 2025, followed by a decrease that lasted 10-25 years, and a subsequent increase that would either continue until the end of the simulation, level off, or lead to an eventual decrease. Recreationalists and conservationists managing nemoral broadleaf forests showed somewhat similarly-shaped trajectories, while passive owners managing these forests differed, being stable in number until 2065-2080 and decreasing thereafter. There were very few owners managing boreal broadleaf forests, with 2010 numbers ranging from 1-19 depending on the owner type, and results tended to diverge with time (Fig. D.1). Generally, farmers showed increasing trajectories during the first half of the century, followed by more divergence.


Productionist Pine
Productionist Spruce
Productionist Pine-Spruce
Multi-objective Pine-Spruce
Multi-objective Pine-Boreal Br.
Multi-objective Spruce-Boreal Br.
Multi-objective Boreal Br.
Recreationalist Pine-Spruce
Number of Agents
Year
Scenarios
Reference
SSP1-RCP4.5
SSP3-RCP4.5
SSP4-RCP4.5
SSP3-RCP8.5
SSP5-RCP8.5

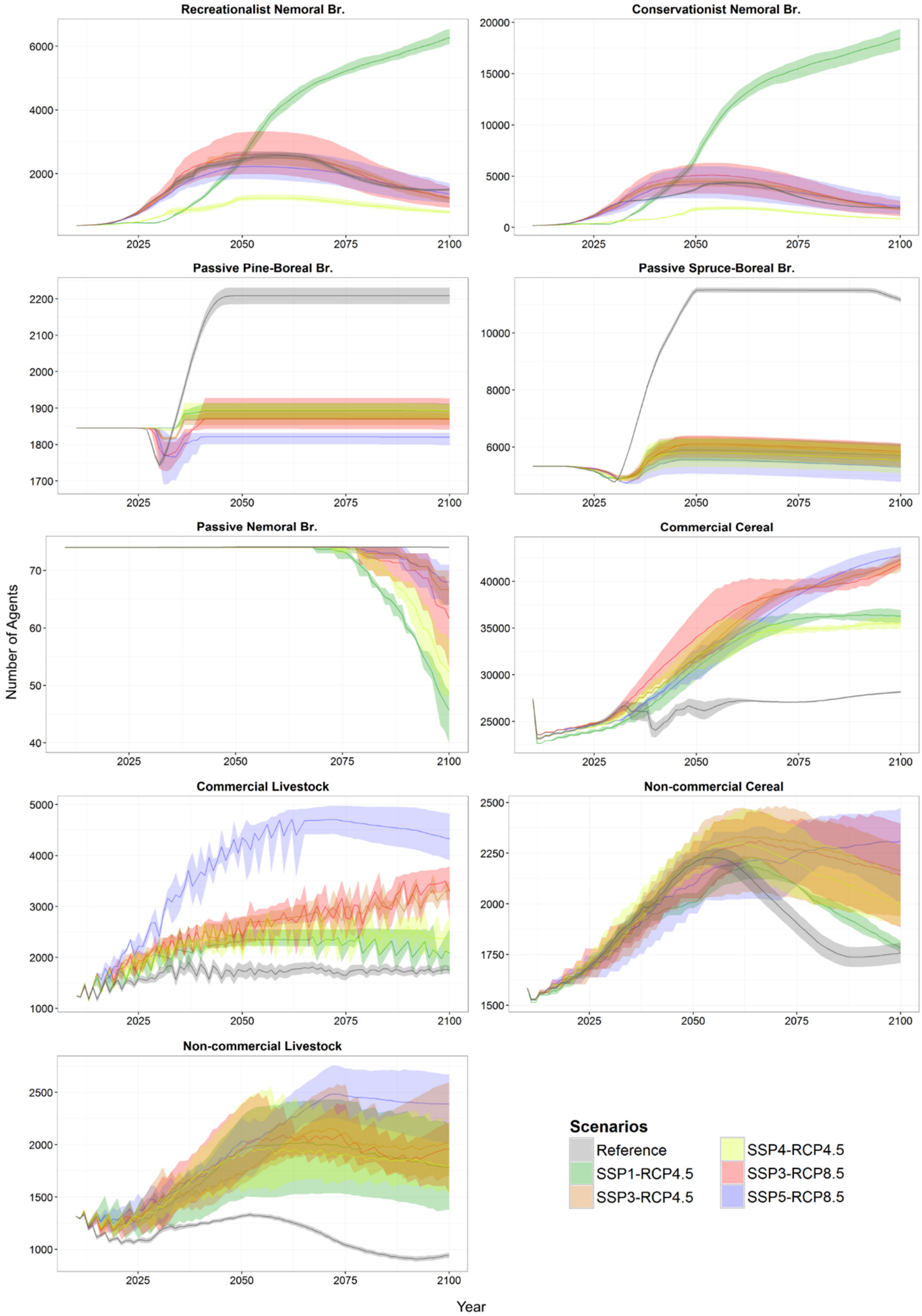


Fig. 12 Ranges in the number of agents for each management strategy through time, for the three climate models (shaded areas) and their means (solid lines) for the five SSP-RCP combinations, and the Reference scenario (mean of 32 variations of random seed). Management strategies implemented by ten agents or less throughout the simulations are displayed in Fig. D.1

Most forest owner types increased in number under SSP1-RCP4.5, followed by SSP4-RCP4.5, while commercial farmers became les numerous under these scenarios. The results for SSP3-RCP4.5 and SSP3-RCP8.5 tend to cluster, as do the results for SSP5-RCP8.5, to a lesser extent. Productionist pine and multi-objective pine boreal broadleaf owners only maintained numbers throughout the simulation under SSP1-RCP4.5, while productionist spruce owners increased in all scenarios except SSP1-RCP4.5. Productionist pine-spruce, however, only increased under the reference scenario. Multi-objective owners managing pine-spruce and spruce-boreal broadleaf forests always achieved higher numbers in 2100 than 2010 despite an initial decrease in numbers for most scenarios. Passive owners managing pine-boreal broadleaf and spruce boreal broadleaf rebounded from their initial decrease approximately 15 years earlier than multi-objective owners managing the same forest types. These passive owners eventually increased numbers towards the end of the century compared with 2010 under all scenarios (except SSP5-RCP8.5 for passive pine-boreal broadleaf), with maximum numbers under the reference scenario. Similarly, recreationalist pine-spruce owners recovered from an initial large decrease 15-20 years earlier than productionist and multi-objective owners of the same forest type, although never reaching their 2010 numbers. Recreationalist and conservationist nemoral broadleaf, in contrast, had higher numbers at the end of the century under all scenarios, especially SSP1-RCP4.5, while the opposite was true of passive nemoral broadleaf owners.

3.2 Ability to cope with global change

I find substantial variability in the coping ability of forest owners through time (Fig. 13). Generally, productionist and multi-objective owners have a higher coping index over longer time periods than other owner types. Nevertheless, both had lower levels of coping until approximately 2025, followed by high levels until the 2060s and a decrease to medium-low levels thereafter. During their final decline in coping ability, levels dropped slightly less for multi-objective owners than for productionists, and grew again slightly during the final decade. Coping levels of recreationalists and passive owners increased until approximately 2030, with coping ability then falling gradually for recreationalists and abruptly for passive owners. Finally, conservationist coping fell to medium-low levels during the first third of the simulation, recovering thereafter and remaining at medium-levels.

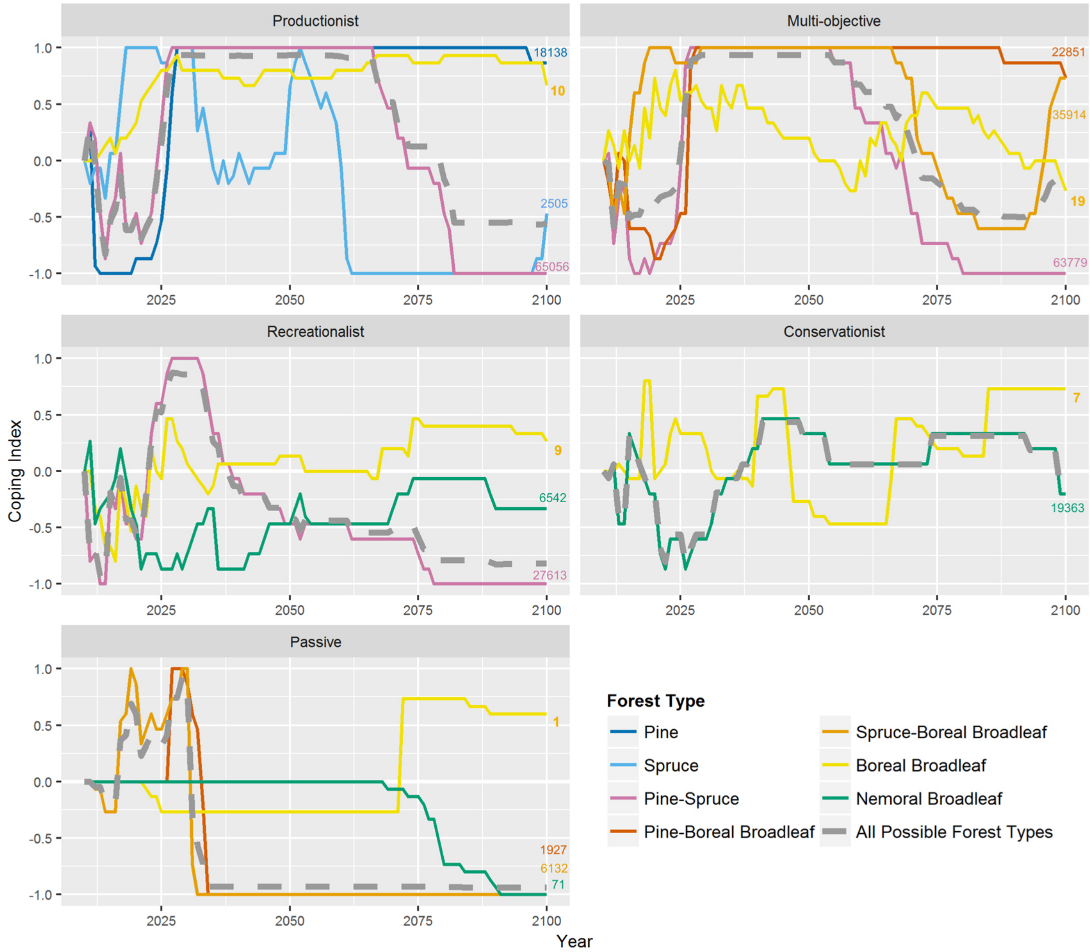


Fig. 13 Evolution of the forest owner type coping index through time. Each panel shows the coping levels of a forest owner functional role in managing different forest types (i.e. the coping ability of a forest owner under different management strategies) (solid lines), and the mean coping levels of each functional role weighted by the number of agents of that role managing each forest type each year (dashed lines). The number of agents implementing each management strategy in 2100 is shown to the right of each solid line

These dynamics differed significantly across forest types. For instance, pine-spruce owners always experienced large decreases in coping levels from a peak in the 2020s, whatever their objectives. Pine, pine-boreal broadleaf, and boreal broadleaf forest owners were more robust, having medium-to-high levels of coping ability from the mid-2020s onwards, with the notable exception of passive owners of pine-boreal broadleaf forests. Nemoral forest owners had medium-to-low coping abilities throughout, with passive owners again faring the worst.

4 Discussion

The results demonstrate that the competitiveness and coping of a management strategy is informative about the coping capacity of individuals implementing that strategy, and ultimately about the strategies' suitability for climate change adaptation. The approach accounts for a range of ES provided through the implementation of different forms of forest and farmland management, and so goes beyond simple yield or economic-based assessments. It also allows an exploration of the potential future development in ES supply to assess the overall capacity of the forest sector to adapt to climate change. Our modelling approach has nevertheless some limitations, which are discussed in Chapter 3.

The results suggest that substantial forest felling will occur in Sweden prior to 2050 as a consequence of the legacy of past forest planting in Sweden (Chapter 3). The felling contributes to the decline during this period in the numbers of owner types managing conifer or conifer-broadleaf forests. These rapid changes demonstrate the importance of large-scale felling events as windows of opportunity to incorporate new forest management strategies for adaptation to a changing global context. Hence, institutions such as national or regional governments, supra-national institutions or owner associations that administer top-down measures (e.g. information dissemination, awareness creation) can maximise their ability to trigger effective adaptation by targeting these periods. However, this requires advance planning as well as avoidance of new waves of planting that increase vulnerability in the supply of ES associated with uneven age distributions (Chapter 3; Rammer and Seidl 2015).

After the initial felling period, higher rates of forest planting and competition for land occur, which are more strongly dependent on the climatic and socio-economic change assumptions. During this later period, the majority of forest owner types were most competitive under the SSP1-RCP4.5 scenario largely due to the greater demands for forest services, the higher importance attached to meeting these demands, and lower demands for food. For similar reasons, commercial cereal and commercial livestock farmers were much more competitive under SSP3-RCP4.5, SSP3-RCP8.5 and SSP5-RCP8.5. The similarity between the agent types in SSP3-RCP4.5 and SSP3-RCP8.5 indicate a relatively low impact of climate change on forest owner competitiveness compared to that of behavioural attributes or societal demands. This is largely consistent with previous research suggesting that future socio-economic conditions are more important than climate change for land-based sectors (Brown et al. 2015; Brown et al. 2016c; Harrison et al. 2015; Harrison et al. 2016). Nevertheless, in many cases uncertainty ranges for the number of agents were substantially larger for RCP8.5 than for RCP4.5, which arises from the larger differences in productivity between climate models in RCP8.5 than in RCP4.5.

That boreal broadleaf forest owner numbers remained very low throughout all simulations suggests that planting boreal broadleaf forests alone (i.e. not in combination with other forest types) is not a good way of adapting to future global change in Sweden. However, the low numbers of such forests in all scenarios means that few other conclusions can be drawn regarding owners of these forests. In contrast, recreationalists and conservationists managing nemoral broadleaf forests experienced substantial growth in numbers in spite of starting at relatively low initial numbers (371 and 180 respectively). Their substantial increase was mainly caused by the gap between supply and demand for nemoral broadleaf timber being relatively large throughout the simulations (Chapter 3). Furthermore, the high importance attributed to meeting demands for carbon storage, biodiversity and recreation in SSP1-RCP4.5 relative to other scenarios made these owner types considerably more competitive in SSP1-RCP4.5. Taken together, these results demonstrate the considerable importance of changes in the demand and supply levels of forest ES in determining future land use change; a system component that is rarely considered in the current generation of forest sector models.

Most owner types whose forests underwent major felling during the first half of the century experienced lower rates of felling in SSP1-RCP4.5, followed by SSP4-RCP4.5, despite demands for timber being larger in these scenarios. This is mainly explained by higher demands for carbon storage, biodiversity and recreation, which are supplied at higher levels in older forests, under these scenarios, and by the relatively high importance given to meeting demands for these services compared to demands for timber. Large scale felling events concentrated within a short time period can determine the competitiveness of owner types such as productionist pine or multi-objective pine-boreal broadleaf under different scenarios. Others (for example pine-spruce forest owners in SSP1-RCP4.5), can have a higher competitiveness during the felling period, but become less competitive when the competition process is more intense, indicating the context-dependency of competitiveness.

The large variability in coping ability of most forest management strategies throughout the simulations was also caused by contextual factors such as the magnitude of past large-scale felling events, the magnitude of unmet demand for forest services, or the intensity of competition. Once again, these results indicate that both coping ability and competitiveness are dependent on past events as well as the current environmental and socio-economic situation. This suggests that the suitability of management strategies for autonomous adaptation is not an inherent, static characteristic of the system, but is dependent on the change in socio-environmental interactions through time.

Furthermore, the success of different management strategies is dependent on spatial characteristics (e.g. land productivity, infrastructure), as illustrated by regional differences (across Sweden) in changes in management strategies throughout the simulations (see Chapter 3).Although the impact of climate change on productivity did not substantially affect the competitiveness of management strategies in this study, it is likely that climatic change would determine the appropriateness of strategies in other areas of the world (e.g. Gauthier et al. 2014; Hannah et al. 2011; Pardos et al. 2015; Temperli et al. 2012). Thus, I can conclude that the suitability of strategies for autonomous adaptation is a dynamic trait of the system that is dependent on spatio-temporal contexts. Consequently, the implementation of adaptive management throughout stand rotation periods (i.e. monitoring and adjusting silvicultural operations such as thinning) (Pukkala and Kellomaki 2012), may contribute to the successful adaptation of forests to future global change.

Moreover, the dynamic nature of the suitability of management strategies for autonomous adaptation indicates the need to re-evaluate the approaches currently used to study the adaptive/ coping capacity of individuals and their management strategies in socio-ecological systems. This is especially so in sectors such as forestry, where the decisions of owners and the implementation of strategies are time-decoupled from the provision of ES. Process-based models may therefore be a more appropriate method of studying autonomous adaptation and future adaptive/ coping capacity than models that project these capacities using static indicators based on discrete time snapshots.

The management strategies of multi-objective owners were found to be more suitable for adaptation than those of owners with different objectives managing similar forest types. While managing forests for multiple objectives often implies both synergies and trade-offs between the ES provided (Chapter 3; Gamfeldt et al. 2013; Holm 2015), the success of multi-objective owners reflected a higher capacity to provide, under the simulated scenarios, more benefits through the provision of multiple services than owners with less diverse objectives.

This finding is consistent with that of Brown et al. (2016c), which identifies the ability of multifunctional management to provide different services and limit trade-offs as very important in shaping land-use change across sectors. This supports the adoption of multiple objective or multifunctional forestry, which has been integrated into national forest management planning in many countries over the last decades (e.g. Carvalho-Ribeiro et al. 2010; Rico and Gonzalez 2015; Swedish Forest Agency 2014c; Zhang et al. 2000). It is still necessary however to better understand the extent to which multiple services can in reality be produced together, and multifunctional management needs to be better incorporated in models of the forestry sector (e.g. by incorporating more comprehensive sets of ES). Owners with a narrower set of objectives were, however, also somewhat successful at using particular management strategies (e.g. productionist spruce), and can be particularly successful in particular locations and scenarios. Therefore, while multi-objective management strategies may be a good solution to meet present and future forest service demands, it does not represent a panacea for all future conditions and is unlikely to prove superior to the maintenance of heterogeneity in ownership and management objectives across larger scales (Rammer and Seidl 2015). Hence, a combination of strategies that can support multifunctionality at different scales is advisable.

I found that while some management strategies may become more competitive in the future (e.g. multi-objective spruce-boreal broadleaf), and especially so for particular scenarios (e.g. SSP1-RCP4.5), others may become less competitive, such as productionist pine in SSP3-RCP8.5. Nevertheless, some strategies that are less competitive now may still become more competitive under future conditions , as illustrated by productionist pine or multi-objective pine-boreal broadleaf. This suggests that, while competitiveness and coping ability are both determinants of the suitability of a management strategy for autonomous adaptation, they are independent of one another and provide complementary insight into the adaptation process. This suggests the need to evaluate these concepts separately in studies of coping and adaptation, which has not to date been identified in the adaptation literature.

The objectives of forest owners made a difference to both the competitiveness and coping ability of forest types. It is therefore important to not only consider the suitability of tree species or species compositions in adapting to climatic or socio-economic change, but also how well management practices associated with particular objectives may perform under future changes. Species choice and composition is not an adaptation solution on its own.

Hence, policy makers and forest advisors need to understand that societal demands for ES will be most successfully met when forest types are managed in ways that align with owner objectives.

Besides age constraints on when forests could be substituted or management strategies changed, I assumed free conversion from one management strategy to another (e.g. productionist to conservationist), competitiveness allowing. Such a conceptualisation may be correct if the change in strategy is associated with a change in ownership. However, in terms of actually realising adaptation by individual or collective owners, there may be barriers to transitioning to different strategies due to knowledge, values, attitudes and objectives (Boon et al. 2010; Kilgore et al. 2008; Kline et al. 2000a). In this case, the change from a strategy based on a particular set of objectives to one based on different objectives is likely to be more gradual than simulated here, and in many cases may not happen at all due to the reluctance to alter objectives. Furthermore, changes between strategies with more similar sets of objectives (e.g. productionist to multi-objective) are more likely than changes to more distinct strategies (e.g. productionist to conservationist). Governmental organisations have therefore an important role in promoting successful adaptation strategies and facilitating, as far as possible, transitions between strategies with very different objectives when appropriate.

Even where successful individual-level adaptation is achieved, successful sectoral-level adaptation does not necessarily follow. Our results indicate that under enhanced competition and high rates of forest land-use change (leading to shorter rotation periods), the Swedish forestry sector may not be able to meet demands for ES (see also Chapter 3). This indicates that even if potentially successful strategies are available and implemented as adaptations, contextual conditions may affect their impact in the forestry sector as a whole and the extent to which societal demands for forest services are met. Hence, planned adaptation via policies and incentives is likely to be essential in addition to autonomous adaptation carried out by individual owners, if future ES demands are to be met. Our findings suggest that process-based models that account for dynamism in competitiveness and coping ability, ES provision and forest owner decision-making have a crucial role to play in supporting such adaptation measures.

5 Conclusions

The suitability of management strategies for adaptation is dependent on spatio-temporal dynamics. Adaptation is not a static, inherent characteristic of a system, but evolves in response to changing contexts that include both the external global change drivers and, importantly, the internal dynamics of agent interactions. Process-based models, capable of simulating the heterogeneity of real-world actors, socio-environmental interactions and change, are therefore a valuable tool in studying adaptation processes, including future coping and adaptive capacity. Competitiveness and coping ability were shown to be independent determinants of the suitability of management strategies for adaptation, each providing different, but complementary types of information about the adaptation process. Furthermore, assessment of the extent to which a diverse range of ES can be provided by mono- or multi-functional management is critical in understanding the potential of future adaptation strategies.

The objectives of forest owners determine the success of particular species and species combinations in adapting to global change. Among the management strategies simulated, those aiming to provide multiple ES in a balanced way may be better at adapting to global change in regions with a bio-physical and socio-economic context such as Sweden. Across large regions, a combination of management strategies is nevertheless advisable to meet forest service demands by taking advantage of location-specific context and changes, and to present response heterogeneity in the face of uncertain global change. Many findings (e.g. the competitiveness of management strategies) were more uncertain under high-end climate change scenarios, meaning that more extreme futures will be more difficult to adapt to. Large-scale felling events represent windows of opportunity to incorporate new forest management strategies for sectoral adaptation in a changing global context. While management strategies exist that are suitable to successfully cope with future global change, their implementation does not guarantee that the forestry sector at the larger scale will be able to adapt successfully. Forestry in the future will likely be unable to meet societal demands for forest services on the basis of autonomous adaptation alone. Therefore, top-down mechanisms such as monitoring, providing information about potentially more successful strategies, and promoting proactive decision-making are necessary to help individuals and the sector as a whole to meet ES supply goals.

Chapter 5

A conceptual model of environmental institutions and their adaptive actions in the Swedish forestry sector

This chapter presents a conceptual model of institutional actions in socio-ecological systems that can be applied in the future to agent-based models as presented in Chapters 2 and 3.

1 Introduction

Interconnected market forces and the rise of international environmental conventions lead institutional drivers to affect land-use from local to global scales (Lambin et al. 2003). Understanding how institutions affect the complex systems within which they operate (Pierson 2000) requires the identification of mediating factors and adaptive strategies at operational levels as well as insight into the interplay with individuals and other institutions across spatial and temporal scales. While formally an institution can be described as an underlying pattern or rule of behaviour in a society, the term is commonly used in the environmental science literature to refer to a specific organisation, a policy instrument or a policy programme (Dovers and Hezri 2010). In this paper, I use the term institutions to refer to formal organisations.

In socio-ecological systems (SESs), within which land managers and institutions interact with the bio-physical environment, complex institutional systems can be distinguished (Ostrom et al. 1999; Pierson 2000). Multilevel governance can be understood as the result of institutions, including NGOs and companies, acting at multiple levels (e.g. municipality, state, supranational) (Yang et al. 2015). Their actions and interactions constitute decision-making systems in environmental matters such as land use or natural resource management (Keskitalo 2013). While not all institutions involved in a socio-environmental system coordinate or are even aware of other institutions' decisions and actions, it is the joint effect of their actions that determines how the system evolves.

To understand changes in land use and natural resource management, it is important to understand institutions and their actions and interactions with one another and with individual decision makers (e.g. land owners) (Agrawal and Yadama 1997; Ostrom 2005). Institutions have almost exclusively been dealt with in a qualitative way in the environmental science literature. Even though SESs are often explored using models, attempts to model institutions explicitly are rare (Campo et al. 2009; Purnomo et al. 2005; Wang et al. 2013). In practice, SES models usually consider institutional policies and interventions as exogenous factors that differ between scenarios, but which are not included endogenously (i.e. represented as processes) within the model (e.g. Guzy et al. 2008; Ralha et al. 2013; Van Berkel and Verburg 2012). This approach can be useful in studies of the disparities between the outcomes of various policies, but is not satisfactory in representing the system feedbacks arising from institutional decision making. Moreover, institutions are made up of people with inherent behavioural mechanisms and decision-making processes that underpin agency.

Models capable of simulating human behaviour and decision making, such as agent-based models (see Chapters 3 and 4; (Ferber 1999), are becoming increasingly popular within the SES modelling community (Rounsevell et al. 2012). These models, however, tend to simulate the behaviour of individuals, while the behaviour of collective entities such as organisations has largely been neglected (Rounsevell et al. 2014). Including institutions in these models, explicitly, could allow better understanding of the influence on, and dynamics of, institutions within SESs. While a few studies have undertaken the representation of institutional behaviour in models of, for instance, forestry (Purnomo et al. 2005) or water management (Bohensky 2014; Schluter and Pahl-Wostl 2007), their treatment has been largely context-specific.

Here, I present and demonstrate a generic conceptual model to represent institutional decision-making and interactions to support modelling of institutional behaviour and its impacts beyond specific socio-ecological contexts. I contend that the applicability of this conceptual model across different contexts, and therefore the possibility of using models to explore aspects of institutional decision-making and policy implementation in greater depth, can contribute substantially not only to SES analysis, but also to fine-grained analysis of policy planning.

I develop a generic conceptual model of institutional action applied to SESs based on (a) a (static) conceptual model of institutional types supported by empirical evidence from the Swedish forestry sector, and (b) an overview of institutional decision-making processes with a focus on SESs. I demonstrate the application of this conceptual model by creating an exploratory dynamic model of institutions that interact in a simple forestry governance system and run simulations under three scenarios. On the basis of these results I discuss the advantages and opportunities for further development of the conceptual model and its components, the possibilities for institutional model parameterisation, the benefits of coupling institutional models with other SES models that simulate human-environment interactions, and the usefulness of such models in policy-making and planning.

2 Conceptual background

Institutional actions result from external drivers and internal coordination mechanisms (Lengnick-Hall and Beck 2005; Ostrom 2005). External drivers are found in the environment within which an institution operates. They can relate to the bio-physical environment (e.g. climatic change) or to the actions of other actors in the system (e.g. lobbying by other institutions). Internal coordination mechanisms are the 'cognitive' processes (and automatisms that do not require any thinking) that underpin the decision-making of an institution in response to external drivers. Below, I describe how an institution makes decisions and implements these in the context of land-use change steered by its internal mechanisms and external drivers.

2.1 Ecosystem service provision

Institutions participating in SES governance either directly or indirectly affect the provision of public (e.g. biodiversity, storm protection) and private (e.g. timber, food crops) goods and services known as ecosystem services (ES) (Millennium Ecosystem Assessment 2005). ES are the benefits provided to humankind by ecosystems. Incorporating ES into a conceptual model provides an appropriate bridge between the provision of benefits generated by ecosystems and the demand for those benefits that institutions aim to fulfil. The goals pursued by institutions are underpinned by the ES that they seek to affect. Therefore, an institutional goal can be defined as a cluster of the (more specific) objectives for ES provision. For instance, the goal of achieving forest multi-functionality can be broken down into a number of balanced objectives for forest service provision (e.g. timber, biodiversity, carbon sequestration).

2.2 Short term versus long term objectives

Formal institutional actors, especially governments, are concerned with the short-term consequences of their actions; long-term effects tend to be heavily discounted because of electoral cycles and other considerations (Pierson 2000). Long-term consequences tend only to become politically salient when they have little effect on short-term elections. In the case of forestry, however, long-term objectives are important because of the requirement for long-term planning, and many of the implications of institutional decisions only play out in the long run. The long growth periods of trees and the resulting slow generation of ES and revenues, which are largely dependent on forest stand age, mean that outcomes from a policy related to changes in tree species or in management strategies that is implemented today may only be expected to have an effect after several years or decades. Non-governmental institutions such as environmental NGOs or land-owner associations may establish their long-term objectives according to a similar rationale. In the short-term, they need to set their priorities according to the levels of ES they aim to achieve.

2.3 Actions

Institutions place constraints on, and create opportunities for, other actors (e.g. restricting or allowing access to land, labour, financial capital, technology, or information) through various actions, including the implementation of regulations and norms (Batterbury and Bebbington 1999; Morgan et al. 2010). They therefore choose those actions that are (or appear) most effective in achieving their objectives. In turn, individuals and organisations respond to institutional actions and to the environments they create (North 1990; Pierson 2000).

2.4 Institutional adaptation

Institutional adaptation typically consists of a sequence of events, including (1) the experience of environmental change and uncertainty, (2) the execution of institutional actions to deal with these changes and uncertainties, and (3) the realization of institutional outcomes and performance consequences (Lengnick-Hall and Beck 2005). Both environmental conditions and organisational capabilities shape an institution’s response and the consequences of these responses.

Chakravarthy (1982) presented a framework of *adaptive fit* that includes three states: unstable fit, stable fit, and neutral fit. Each of these states can lead to institutional endurance through adaptation. The term *adaptive fit* is used to mean that an institution is able to accommodate the new environmental state.

A defensive strategy (i.e. based on reducing an institution's interactions with its environment) promotes *unstable fit*. Institutions with unstable fit remain vulnerable to external events and depend on buffers for protection from the possible adverse consequences of environmental change. Unstable fit is best suited to an environment that changes slowly and predictably. In forestry systems however, a defensive strategy is not sufficient because it requires long-term planning despite slow processes.

A reactive strategy can support *stable fit*. Institutions with stable fit attempt to identify and react to environmental change by realigning the institutional decisions as environmental conditions change. This form of adaptive fit is appropriate for institutions confronting moderate levels of environmental complexity. Stable fit allows for the process known in climate change science as *coping* or *autonomous adaptation* (IPCC 2012; IPCC 2014b), which involves addressing adverse conditions with the aim of achieving functioning in the short to medium term.

A proactive strategy can promote *neutral fit*. Highly complex environments require neutral fit so that institutions are able to reduce their vulnerability to change by anticipating external shifts. This type of adaptive fit allows for *anticipatory adaptation* processes (IPCC 2014b). Anticipatory adaptation takes place before the impacts of environmental change are observed.

A determinant of institutional adaptation is learning, since institutions, as with individuals, learn from experience (Pierson 2000). In the context of land use decision-making, the International Institute for Environment and Development (1994, as cited in Kengen 1997) stated that, in practice, there is no alternative to presenting the policy-maker with a range of indicators and models. Surveying and monitoring enables institutions to learn about the state of the bio-physical environment and its associated services. Information gathered through these methods is however often imperfect (Monk 2014; Yokomizo et al. 2014). In turn, institutions learn from such information and make decisions limited by uncertainty (Bryson et al. 2010).

Also, measuring the performance of actions can allow institutions to learn about their effectiveness. However, it is often very hard to observe or measure important aspects of performance (Pierson 2000). Finding reliable indicators of performance is difficult because the outcomes of the actions of an institution are highly dependent upon the actions of others and the nature of the environmental change. Thus, outcomes cannot be attributed solely to the actions of one institution. Even if mistakes or failures in policies are apparent, improvement through trial-and-error is difficult and time-consuming. Therefore successful learning is difficult: while learning does happen, it is often underpinned by inaccurate or incomplete information.

3. Overview of methods

3.1 Conceptualising institutional agent types and their interactions

Information about Swedish forestry institutional structures, actions and interactions with other institutions and with forest owners were derived from peer-reviewed and grey literature. This knowledgebase was used to create an institutional network representing a (static) conceptual model of interacting forestry institutions. I created a typology (Rounsevell et al. 2012) of institutions to deal with institutional heterogeneity and reduce the large number of institutions involved in the system to a set of similar “types”. Generic institutional types were identified through descriptors (e.g. ‘research supplier’, ‘forest campaigner’, ‘government’) that were commonly used in the forest governance literature (e.g. Meidinger 2006; Werland 2009). Because of their specific roles, each Swedish forestry institution was found to correspond to a generic institutional type, *viz* supranational institutions, certification programmes, national, regional and local governments, environmental NGOs, owner associations and research suppliers. Generic goals and actions of institutional types were identified from the goals and actions of the corresponding real institution and their actions in achieving their goals.

3.2 Decision making and the institutional action conceptual model

Supranational institutions, environmental NGOs, owner associations and government, which have goals and actions that relate directly to the different ES that land can provide, were modelled as agents (see Rounsevell et al. 2012). For simplicity, research suppliers were represented implicitly in terms of (the quality of) information about the environment that other institutions use to inform their decisions (i.e. they are not attributed agency).

I then defined and incorporated decision-making processes into each institutional type. An institutional action conceptual model was adapted from the generic LARA framework (Lightweight Architecture for boundedly Rational Agents) (Briegel et al. 2012) since it is able to account for institutional decision-making processes. The core elements of LARA for deliberative decision-making are (a) a decision-maker's *goal preferences*, (b) a set of *behavioural options* (BO), and (c) some *knowledge about the utility of each BO to achieve the different goals*. The decision-maker's selection of a BO is based on its beliefs about the utilities of BOs in the present situation, obtained by multiplying situational goal preference weights with the utility of each BO in achieving each goal. A situational goal preference results from multiplying a basic preference with the situational effect (weight) on that preference. In our conceptual model, LARA's core elements for deliberative decision-making were reframed to the context dealt with here and termed *service preferences*, *potential actions* and *perceived effectiveness of potential actions*[2].

The model's path of information processing and decision-making works as follows: the information perceived by an agent from the environment is stored in its memory, together with its goals, service preferences (understood as the relative importance attributed to objectives), and potential actions. Goals correspond here to levels of service provision that an institutional type wishes to achieve. Given that the optimum production level of a service is found where it meets the demand, it is reasonable to assume that institutions will try to ensure the fulfilment of self-defined demand levels. Following information storage in memory, the institution deliberates, leading to the establishment of action priorities. Finally, the institution evaluates and decides which action/s to implement.

3.3 Example model illustration: a simple forestry governance system

I illustrate an application of the conceptual model through a simple forestry governance model generated using Microsoft Excel and Visual Basic. The model was parameterised by assigning levels of service preferences and perceived effectiveness of potential actions to reflect the information gathered about the institutional structures of the Swedish forestry

2 Corresponding to *goal preferences*, *behavioural options* and *situational utilities* respectively in the LARA framework.

sector, for three types of institutions: government, environmental NGOs, and owner associations. Levels of service preferences were assigned according to the institutional type goals defined in Table 11 (see results section 4.1). Potential actions of the government are to subsidise service production, to set production quotas, and to invest in infrastructure (e.g. roads). Environmental NGOs and owner associations may lobby the government to support the supply of the different services. Even though governmental actions are meant to affect forest managers' ability to generate services, given the focus of this paper on institutional modelling, managers were not modelled and governmental actions were assumed to influence service provision (and hence supply-demand difference) directly.

Three hypothetical scenarios were generated that differ in the production levels of the three services provided by managed forests at a given moment in time. The competing services are timber, biodiversity, and recreation. Each model simulation was run for 50 time steps.

4. Institutional types, the conceptual model and its application

4.1 Institutional types conceptual model

Appendix E provides a set of narratives describing the actions of, and interactions between, Swedish forestry institutions. Fig. 14 represents these interactions as a network. Given the complexity of this network, and the difficulty of representing and parameterising such great number of institutions and interactions in a simulation model, Fig. 15 was developed as a generic interpretation of Fig. 14, to create a (static) conceptual model of interacting forestry institutional types. The goal of research suppliers is to generate and provide knowledge (Table 11), but this is dependent on the goals of research funders. The goals of a government depend on its electorate, and on the goals of stakeholders who influence the government. The goals of environmental NGOs and owner associations are more stable and reflect the values, attitudes and beliefs of members (Eagly and Chaiken 1993).

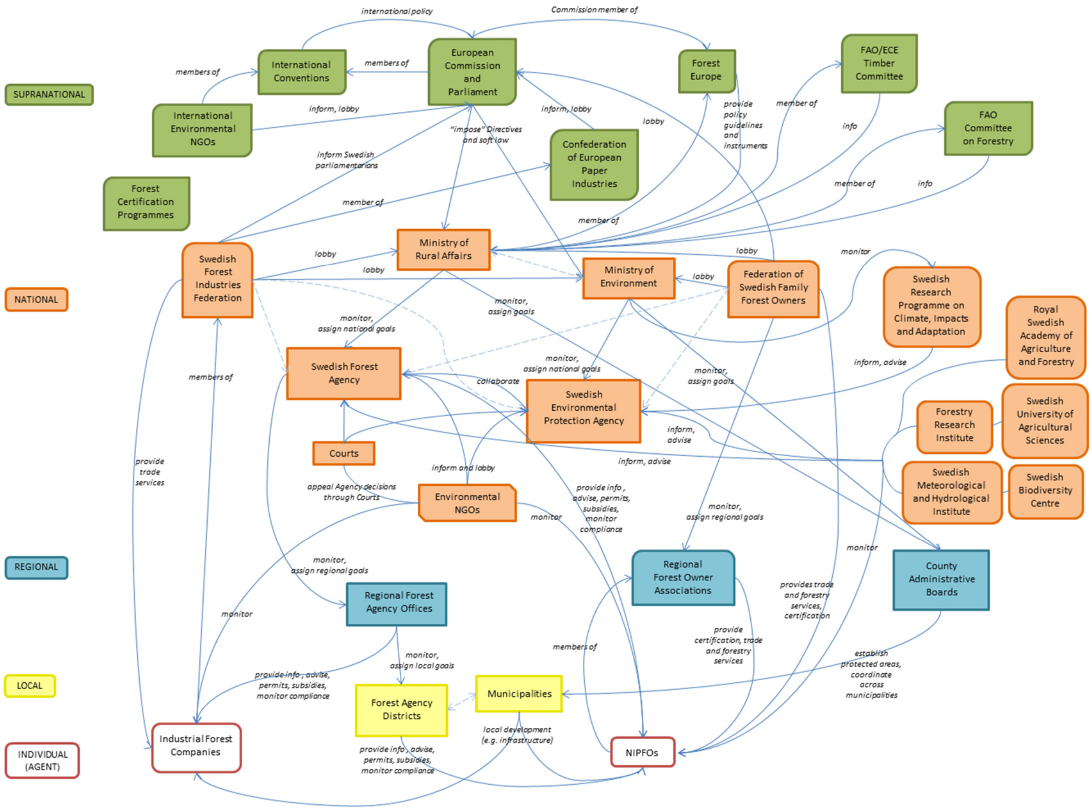


Fig. 14 Representation of the relationships between institutions and forest managers in the Swedish forestry sector based on their actions and the geographical-hierarchical level (colour-coded) at which they operate. Similarly-shaped boxes represent a particular institutional type: a) rectangle = government; b) rounded corners = research suppliers; c) snip diagonal corners = environmental NGOs; d) round upper corners = owner associations; e) round diagonal corners = supranational institutions. All solid arrows are unidirectional. Dashed arrows signify presumed connections that could not be confirmed empirically. 'Forest Certification Programmes' connections are omitted for brevity

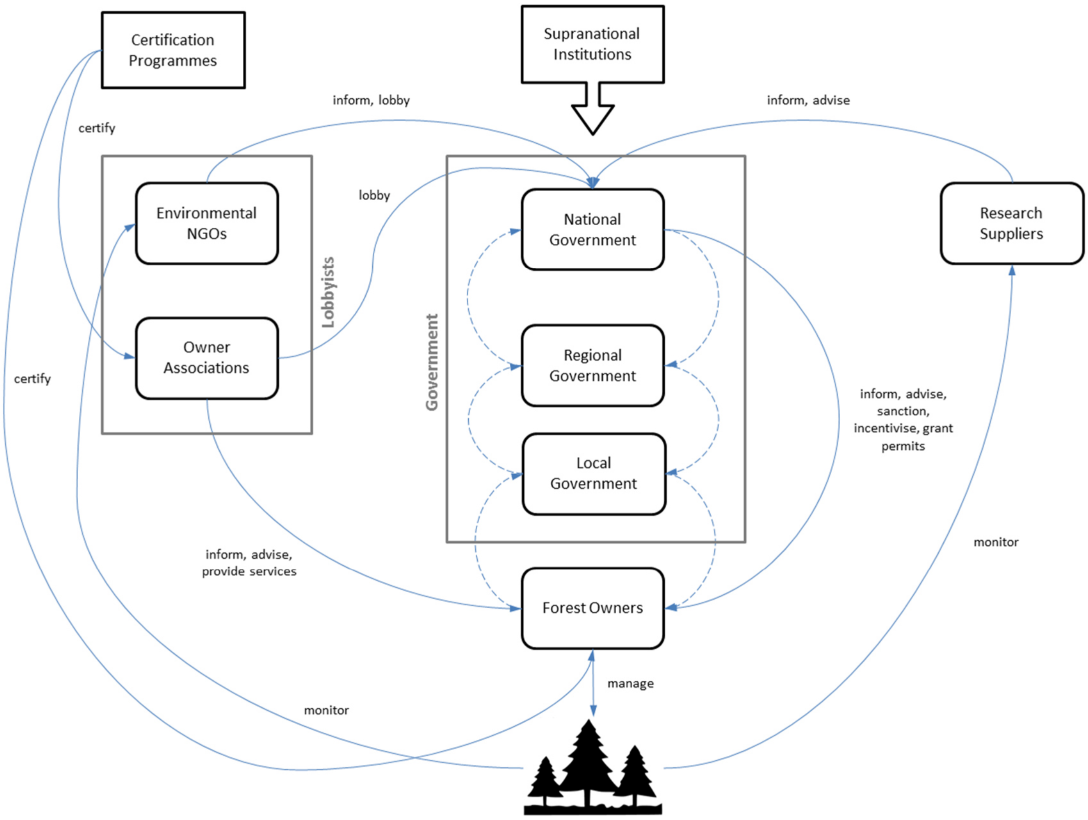


Fig. 15 Conceptual model of generic institutions in the Swedish forestry sector. Arrows represent the direction of influence through actions performed by institutional types

Table 11 Institutional type goals and actions in relation to forestry

Institutional type	Goals	Actions
Government	- Production - Environmental - Climate change mitigation and adaptation - Balance different stakeholder goals	- Inform/advise - Sanction - Incentivise - Grant permits
Research Suppliers	- Provide knowledge (weather, water, climate, impacts and adaptation, forest production, operations, biological natural resources, biodiversity)	- Monitor environment - Inform/advise
Environmental NGOs	- Nature conservation - Sustainability - Climate change mitigation and adaptation	- Monitor environment - Inform - Lobby

Owner Associations	- Owner profitability	- Inform/advise - Provide services - Lobby
Supranational Institutions	- Sustainability - Multi-functionality - Climate change mitigation and adaptation	- Establish international hard and soft law - Inform/advise - Grant certification

4.2 An institutional action conceptual model

Fig. 16 shows an overview of the institutional action conceptual model. When institutional agents make deliberative decisions about their actions, they do so by considering their service preferences, how well the demands for services are being met (from here on 'supply-demand difference' (SDD)) and the perceived effectiveness of their potential actions. These three decision-making components take values for the services considered by the institutional type, which can be normalised for comparability and to reflect differences between values without the need for additional weighting factors. Values for service preferences (P_s), SDDs (S_s) and perceived effectiveness of potential actions ($E_{s,a}$) are taken by a function (Eq. 1) that calculates final values to determine which actions (a) are taken upon which services (s) and which are not, i.e. action priorities ($AP_{s,a}$). An action is implemented upon a service if its $AP_{s,a}$ value is above a predefined threshold.

$$AP_{s,a} = P_s * (-S_s) * E_{s,a} \quad (1)$$

A service preference P_s represents the relative importance given by an institution to the generation of a certain service in relation to the importance attributed to the generation of other services. Even though institutions may have long-term and short-term objectives, and thereby preferences, I only regard one type of preference for the sake of simplicity.

Planned adaptation to environmental change can be incorporated into the model through adjustments of P_s. An institution may decide to adapt after becoming aware that conditions have changed or are expected to change, with the intention of returning to, maintaining, or achieving a desired state (Easterling et al. 2007). Changing conditions may refer to alterations to an institution's circumstances (e.g. markets) (e.g. Mosnier et al. 2014; Nkem et al. 2010) or the physical environment (e.g. land productivity, fire risk) (e.g. IPCC 2014b; Ostry et al.

2010). These may prompt an institutional agent to readjust the relative importance given to a number of services. Changes to P_s values may also be predefined following decisions to adapt established through scenarios.

The service SDD S_s indicates to the institution how well the demand for a service is being met at the time of deliberation. If demand is unmet, then -1< S_s<0, while in a situation of oversupply 1> S_s>0. In Equation 1, S_s is multiplied by -1 because I assume that the magnitude of the intended change in supply, in order to meet the demand, is inversely proportional to the magnitude of the SDD. As information monitored by research suppliers or others is generally imperfect, uncertainty can be incorporated in monitored S_s values through an uncertainty distribution. The resulting value of multiplying P_s and S_s reflects the institutional agents' situational service priorities.

The perceived effectiveness of an institution's potential action to increase or decrease the production of a service $E_{s,a}$ reflects an institution's self-assessed efficacy in increasing/decreasing production of the service by implementing a particular action. Effectiveness would be limited in the real world by factors such as budget restrictions, available knowledge or power. This perceived effectiveness may be influenced by learning. Knowledge acquired by an institution at any point in time that does not relate to service provision may be incorporated via changes in the perceived effectiveness of certain potential actions and preferences for affected services. Such a mechanism would be relevant when simulating processes of planned adaptation to climate change, for instance. New knowledge becoming available about the means for adaptation (e.g. improved policy tools, new technologies) (e.g. Harrison et al. 2013; Lindner et al. 2010) could justify reassessing the institution's perceived effectiveness of relevant actions. Additionally, changes to an institution's power or available (financial) resources may entail a shift in the perceived effectiveness of its potential actions. The $E_{s,a}$ can only take values between 0 and 1 if the action is intended to increase service production, and between 0 and -1 if the action is meant to reduce service production. Adjustments to $E_{s,a}$ values can be introduced at given points in time through scenarios, reflecting for instance increases or decreases in dedicated budgets or new available knowledge.

The action priority $AP_{s,a}$ indicates the degree of priority of an action (i.e. implementation). An action with a priority value higher than the exogenously determined threshold value will be imposed on the corresponding service. Otherwise the action will not be implemented.

Implemented actions are intended to effect land capitals, land-use, institutions, or land managers' ability to produce services. For example, if the action "subsidise" for the service "biodiversity" obtains a value higher than the implementation threshold, then a subsidy to biodiversity will be enacted. As a result of subsidising biodiversity, levels of biodiversity may or may not increase. In the subsequent time step research suppliers will monitor biodiversity levels again and a new decision-making process will commence.

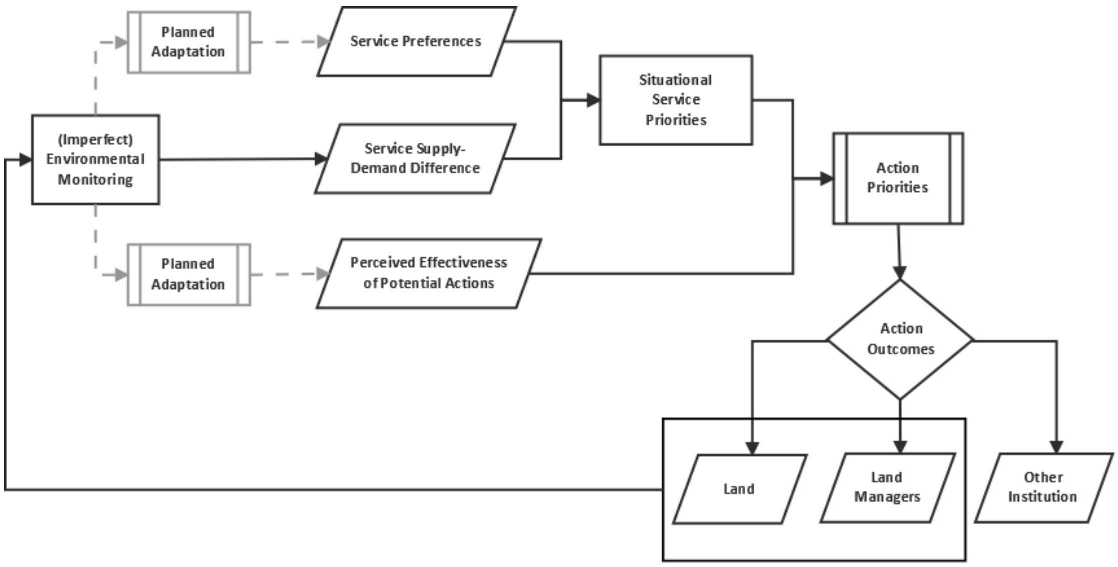


Fig. 16 An overview of the institutional action conceptual model showing the relationship between key components of the model and main entities of a SES (i.e. land, land managers, and other institutions). Grey boxes connected by dashed lines represent processes that may occur only once over several iterations

4.3 Example model illustration

4.3.1 Model and Parameterisation

Developed scenarios are shown in Table 12. Table F.1 shows the initial parameters assigned to the model at time t_0, and the changes to some parameters and generated variables at time t_1. Levels of perceived effectiveness of potential actions, and service preferences of environmental NGOs and owner associations are kept constant throughout the simulation. The chosen implementation threshold value is the same for all institutions and, like the action priority values it acts on, may range between -1 and 1. If NGOs or associations lobby the government for a service, the level of governmental preference for the service increases by 0.1. The effect of subsidising or investing in infrastructure is to increase a service's SDD level by 0.1, while setting quotas decreases service SDD by 0.1. Also, to simulate competition between services supplied from limited land resources, service supply is reduced at each time step if one or both of the other services increase. Equations 2, 3 and 4 illustrate the nature of the changes in timber, biodiversity and recreation respectively subject to the effects of implemented actions and changes in the two other services.

$$S_{T,t+1} = S_{T,t} + EA_{T,t} - \left(EA_{B,t} * C_{B,T}\right) - \left(EA_{R,t} * C_{R,T}\right) \tag{2}$$

$$S_{B,t+1} = S_{B,t} + EA_{B,t} - \left(EA_{T,t} * C_{T,B}\right) - \left(EA_{R,t} * C_{R,B}\right) \tag{3}$$

$$S_{R,t+1} = S_{R,t} + EA_{R,t} + \left(EA_{B,t} * C_{B,R}\right) - \left(EA_{T,t} * C_{T,R}\right) \tag{4}$$

where EA_T, EA_B and EA_R are the effects of the governmental action/s on timber (T), biodiversity (B) and recreation (R), and $C_{i,j}$ is a factor determining the collateral effect of an action targeted at a service (i) on the SDD of another service (j). $C_{i,j}$ parameter values can be found in Appendix F.

Multiple model runs were performed with different initial service SDDs, service preferences, perceived effectiveness of potential actions, and implementation thresholds. A sensitivity analysis of the implementation thresholds was undertaken by measuring the area between a curve of SDD modelled with threshold value 0 and SDD curves with threshold values between -1 and 1 at intervals of 0.1.

Table 12 Descriptions and levels of service supply-demand difference (SDD) for three scenarios used in the example model illustration

Scenario	Description	Service SDD		
		Timber	**Biodiversity**	**Recreation**
TIMBER PROFUSION	Available forest land is managed primarily for timber production while other services are treated as secondary. Timber supply is very high to the point of substantially going beyond the demand. The supply of biodiversity associated with production-oriented forests is low. Under such circumstances some recreation is provided, but it does not meet demand.	0.6	-0.5	-0.1
ENVIRONMENTAL EDEN	A large proportion of the forest land is managed with nature conservation as a primary objective. Supply of timber does not meet demand, while biodiversity is oversupplied. Recreation, being partly associated with levels of biodiversity, is also supplied slightly beyond the demand.	-0.2	0.5	0.1
PERFECT EQUILIBRIUM	Forest land management seeks multi-functionality. Production levels of all three services are equal, but they do not meet the demand.	-0.3	-0.3	-0.3

4.3.2 Modelling Results

It can be observed that, while projected SDDs are somewhat sensitive to changes in all four parameters, they are most sensitive to adjustments in implementation thresholds. While initial SDDs, preferences, and perceived effectiveness of actions have the same weight in determining institutional action priorities, the former is more influential on the outcomes as it sets the starting position for service SDDs.

Sensitivity analysis shows a major effect of implementation thresholds on SDD outcomes. Service SDD projections behave particularly differently if the threshold is above, equal to or below 0 (Fig. 17a). Thresholds above 0 reflect lower institutional intervention, and higher thresholds increase the gap between service provision and demand. Also, after a few initial time steps where institutions may intervene, service provision appears to become ‘good enough’ at which point institutions no longer attempt to influence service provision, which then remains constant over time. If the threshold is equal to 0, intervention becomes intense until, after a few initial time steps, service provision is close to meeting the demand. From this point onward the provision of different services oscillates above and below demand levels, while remaining close to these levels, as a result of service competition for available land and subsequent institutional interventions. If the implementation threshold is set below 0, patterns are somewhat similar to those when the threshold equals 0, except that the oscillations occur around higher SDD values for lower thresholds. This high intervention leads to oversupply as institutions support production even where their action priorities are very low or even negative. Overall, service SDDs are more sensitive to threshold decreases (from threshold = 0) than they are to increases (Fig. 17b).

Simulation runs for the three service SDD scenarios with an implementation threshold of 0.2 generate substantially different SDD projections (Fig. 18). While production levels cause institutional action when they are far away from the demand, in the “Timber Profusion” and “Environmental Eden” scenarios, the service that is initially supplied the most continues to be more supplied than the others, and the least supplied service remains the least supplied. In the “Perfect Equilibrium” scenario, initial provision levels are the same for the three services and they all reach equilibrium after a few time steps at the same level, closer to the demand.

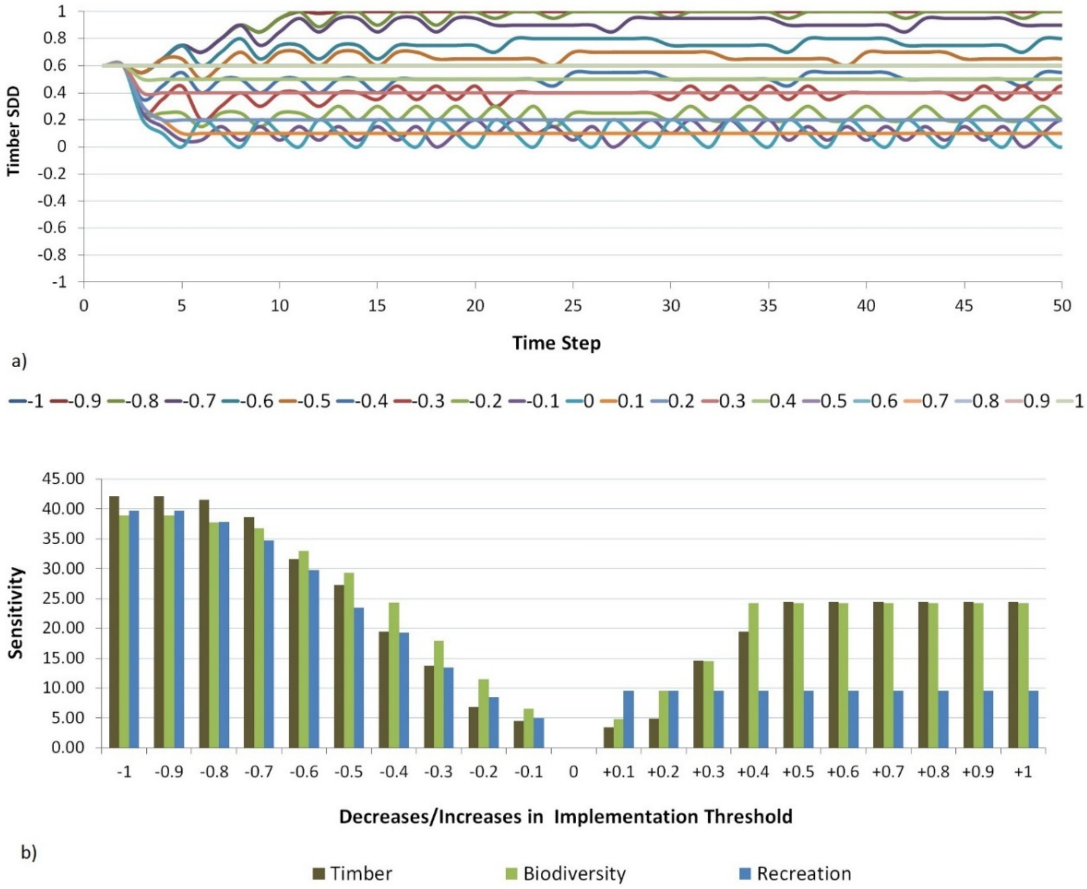


Fig. 17 a) Timber SDD projections for different implementation thresholds between -1 and 1; and b) sensitivity (i.e. area between curves) of service SDD for implementation thresholds with decreasing and increasing values at 0.1 point intervals starting from a threshold value of 0

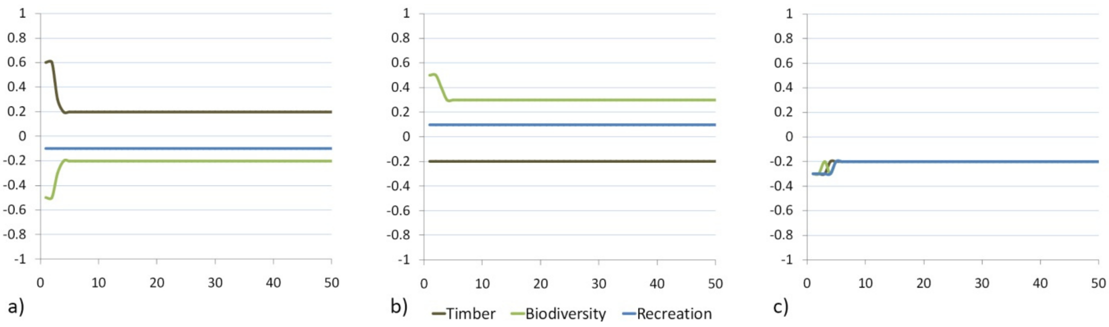


Fig. 18 Timber, biodiversity and recreation SDD projections simulated with an implementation threshold of 0.2 and for scenarios a) Timber Profusion, b) Environmental Eden, and c) Perfect Equilibrium

5 Discussion

The institutional action conceptual model presented here provides an advance in SES models by allowing important institutional dynamics to be represented endogenously. This was demonstrated by a simple, illustrative implementation, but other implementations are possible to address different contexts and a wide range of research questions.

The three institutional types (i.e. government, environmental NGOs, owner associations) can be observed at any level of governance (e.g. local, national, and international) and are likely to occur in all SESs, yet their decision-making parameters may differ. Nevertheless, a model could represent any number of institutional types or more specifically-defined institutions. For instance the interplay could be modelled between a local and a national government, or the interactions between the different organisations involved in the Swedish forestry sector.

While I chose a threshold to select implemented actions, it should be noted that different action-selection mechanisms could be used depending on the decision-making process. For instance, a simple ranking of actions according to action priority levels is an alternative. The implementation threshold reflects the willingness of an institution to intervene in service provision levels. In the simulations presented here equal threshold values were assigned to all institutions for the sake of model simplicity, but a more realistic model may take different threshold values for different institutions and even different services, as actual institutions may not be equally inclined to intervene. For example, in a country such as China, self-identified as a 'socialist market economy', government is more likely to regulate the provision of certain services (e.g. energy) than in a neo-liberal country (Haque 1999; Lo 2013), which would prefer to leave more leeway for the market to self-regulate. Also, a government might be more willing to intervene in the provision of public services such as biodiversity, than in private services such as timber production.

As with many models, the main challenge faced by this conceptual model when developing real world applications is reliable parameterisation (Bravo 2014; Smajgl et al. 2011). In spite of this, however, data collection methods exist that could support model parameterisation; the most appropriate methods being social surveys and focus groups (see Lindlof and Taylor 2011; Nichols 1991; Philip 1998).

In developing an empirical model of institutional types, a common method to parameterise agent types is to use social survey data with a cluster analysis (Chapter 3; Fontaine et al. 2014; Rounsevell et al. 2012). Clustering essentially describes the grouping of subjects (e.g. institutions) based on the similarity of attributes (Rousseeuw 1987). This method can be used as an alternative to the more qualitative way in which institutional types were developed here.

The parameter that represents the perceived effectiveness of potential actions of an institution is the most difficult to define, since it is difficult for an institution to assess the potential effectiveness of its own actions (see section 2.4). Hence parameterisation may not go beyond a 'best guess'. However, this parameter is based on a relative value, so that relative differences in action effectiveness can be considered between actions. Relating the effectiveness of actions to those of others can help in defining the effectiveness levels for actions that are more uncertain. Moreover, this parameter provides a means of exploring the effects of different learning strategies.

In the model illustration presented here, institutions were assumed to be isolated from forest managers and their actions. Nevertheless, institutional models can be useful in studying SESs (e.g. land-use and land-cover change, natural resource management) when coupled with other SES models that simulate interactions between land managers and the environment. Coupling such models would be useful in exploring the feedbacks between institutions, land managers and the environment. A model of land manager agents and their decision-making processes could also be based on the supply of ES providing coherence across the different agent types (Chapter 3).

Models of institutional decision-making in SESs could also inform policy and planning processes. For instance, the conceptual model enables the assessment of the effect of alternative policy instruments on the environment (i.e. institutional actions) or an evaluation of the trade-offs in ES provision resulting from policy intervention (Villamor et al. 2014). Moreover, coupling an institutional model with a land-use model, would allow an evaluation to be made of the impacts of institutional decisions and policies on land-use and land-cover (Matthews et al. 2007) and on the behaviour of land managers. This approach would also support evaluations of the effects of different governance structures on land-use change and ES provision.

6 Conclusions

I demonstrate how the actions and interactions of institutions in SESs can be represented in simulation models. I show that Swedish forestry institutions can be classed into generic categories that support modelling, including parameterisation. I also demonstrate the applicability of the LARA framework in modelling actors in SESs and their decision-making processes, which in spite of representing institutional processes simply, offers insight into system behaviour. The scenario analysis shows that, provided institutional intervention is not high, the relative supply of each ES (in relation to the supply of other services) is not likely to change substantially. Hence, taken in isolation from other external factors of the SES, the overall *status quo* (in terms of relative service supply) is unlikely to change unless institutions intervene strongly. Simulations performed with a lower level of intervention suggest that cooperation between institutions would lead to more consistency in the achievement of policy objectives. By contrast, more competition between institutions causes them to pull in different directions, leading to more variability in service supply.

Complexity and uncertainty are inseparable from policy and planning in SESs. Institutional conceptual modelling has a role in allowing scientists, policymakers and planners to better understand the consequences of institutional actions and interactions on SESs. Such models support better understanding of the key institutional decision-making dynamics, endogenously, in a flexible way across different SESs. This relies, however, on the use of appropriate methods to parameterise institutional service preferences, the perceived effectiveness of potential actions and the willingness to intervene in service provision. The study of appropriate methods to parameterise institutional decision-making in models is therefore an area for further research.

Chapter 6

Thesis Discussion and Conclusions

1 Discussion and conslusions

This study has addressed both adaptation to global change in the forestry sector and the appropriateness of modelling tools such as ABMs to study adaptation processes. Over the four previous chapters, I answered the research questions posed at the introduction of this thesis. In the following two sections, I discuss my findings focusing on first on adaptation and secondly on adaptation assessment and modelling. The limitations of this study and future research are subsequently discussed. Finally, I offer policy recommendations on the basis of findings. Key findings are stated in bold and discussed in each paragraph. Each key finding includes a list of the chapters that contain the relevant literature and evidence supporting it.

1.1 Adaptation to global change

Forestry in the future will likely be unable to meet societal demands for forest services solely on the basis of autonomous adaptation. Reactive adaptation strategies were found to be insufficient to address the effects on service provision of the substantial time-decoupling between the supply of, and demand for, forest services; legacy effects of past land-use change; and competition for land between agents. Even when suitable adaptation strategies may be available to forest owners, successful sectoral-level adaptation will not necessarily follow. Therefore, top-down mechanisms and planned adaptation are necessary to help individuals and the sector as a whole to meet ES supply goals. Such mechanisms could include the promotion of uneven-aged forest management (Lafond et al. 2014; Laiho et al. 2011) and of multi-objective forestry to ensure a more sustained ES provision. Studies like this, help institutions involved in forestry and land use to better inform land owners and other institutions about better/worse adaptation strategies and approaches, so that they will be able to make more informed plans and decisions regarding adaptation. *(Chapters 3 and 4)*

A northward expansion of agriculture and especially of forestry proved positive for both sectors in adapting to changing conditions, under several scenarios, given the substantial land availability and the improved environmental conditions for plant growth. Other studies have suggested the benefits to the provision of forest and agricultural services from a northward expansion in forestry and agriculture due to increased productivities resulting from climate change (Carter et al. 1996; Schoene and Bernier 2012; Yang et al. 2015). Here, forest expansion towards the mountains occurred under all scenarios, reinforcing the argument that climatic conditions may make mountain areas more suitable for forestry due to increasing productivities (Kullman and Kjallgren 2006; Rundqvist et al. 2011; Van Bogaert et al. 2010). Pine-spruce forests managed by multi-objective owners were found to be the most successful in adapting to changing conditions by occupying land at higher latitudinal and altitudinal regions. Spruce-boreal broadleaf forests managed by multi-objective owners were also very successful, but displayed no obvious expansion towards higher altitudes, instead expanding in a more scattered way throughout the country. Their success was mainly rooted in their capacity to generate relatively high amounts of all services compared to forest owners implementing other strategies, especially those with narrower sets of objectives. The capacity to produce a more diverse set of ES, characteristic of multi-objective owners, also makes them more resilient, as they are less dependent on the benefits that each individual service can provide (Lin 2011; Schippers et al. 2015). Additionally, forests with a high degree of heterogeneity in species composition commonly appear less storm-sensitive than monocultures, which helps to spread the risk of storm damage (Jonsson et al. 2015). Therefore, from among the pool of tree species and compositions considered, pine-spruce and spruce-boreal broadleaf forests managed for multiple objectives should be given priority when considering adaptation measures to global change in Sweden. Other strategies may not be as competitive when considering the magnitude of their uptake throughout Sweden. However, the regional differences in biophysical conditions and their distinct evolution through time imply that even the least widespread strategies will be the most successful at particular locations. Therefore, at large scales, policy makers, planners and advisors need to consider a pool of management strategies broad enough to cater to the specificities that occur at smaller scales (e.g. local, estate). *(Chapters 3 and 4)*

Climatic change, ES demands, competition processes, legacy effects of past planting events, and the objectives and behaviour of land owners and environmental institutions, were all drivers of change in land use and ES provision. However, they proved to be differently important and to have different impacts on the system.

Legacy effects of past land-use change can have great impact on future land-use change and adaptation processes, especially in forestry. The national uneven forest age distribution will have a negative effect on the provision of ecosystem services (ES) following the inevitable felling of a substantial proportion of national forests during the first half of this century.

Rounsevell and Reay (2009) also observed negative legacy effects from concentrating forest planting within a relatively short period of time in the UK, leading to very unstable net carbon removal through time. To avoid similar legacy effects in the future, sectoral adaptation should include the spreading of forest planting throughout relatively long time periods (e.g. two or three decades), to ensure that their felling in the future occurs more gradually in time. This can be achieved through uneven-aged forest management, which can provide a more sustained ES provision (including timber yields) over time, and contribute resistance to storm damage due to structural diversity (Kuuluvainen et al. 2012; Lafond et al. 2014; Laiho et al. 2011; O'Hara 2006). Uneven-aged forest management can also build resilience due to a smaller degree of variation in structure over time that allows a greater ability to rapidly return to a pre-disturbance state (O'Hara 2006). While initial delays in planting may have economic and social costs, they should contribute important benefits in terms of a more sustained provision of ES. Also, in the mid and long-term uneven-aged forestry can be just as economically competitive as even-aged forestry (Kuuluvainen et al. 2012). In Sweden, even-aged forestry is dominant (Kuuluvainen et al. 2012). If even-aged forestry remains the preferred strategy at the national scale, the planting of stands should be delayed at some locations. To minimise economic and social impacts on land where replanting is delayed, upon felling land could be put to a different use for some time. Land uses that can provide services within shorter time frames, such as agriculture, may be appropriate for this purpose. However, implementing such an approach would require in Sweden substantial institutional coordination at the national level and policy change. For instance, the current policy preventing the conversion of forested land to agriculture would need to be modified. *(Chapter 3)*

A higher competition for land may lead to shorter forest rotation times. Even though the increase in productivities under several scenarios caused rotations to become shorter, their effect on rotation length was negligible compared to that of competition. In line with the qualitative results of Roberge et al. (2016), I showed that shorter rotations can negatively affect the provision of non-timber ES. In Sweden, rotations that were shorter by 10-15 years in Norway spruce-dominated forest (where typical rotations are 60-80 years) are currently being discussed to decrease the risk of storm and root rot damage (Roberge et al. 2016). If such an adaptation mechanism was broadly promoted, measures aiming to enhance the production of biodiversity and recreation, and carbon sequestration, such as extended rotations and set-asides for conservation (Koskela et al. 2007; Monkkonen et al. 2009), will likely be necessary in order to compensate for the loss associated with shorter rotation times. *(Chapter 3)*

Socio-economic change and land owner behavioural differences were found to have a higher impact on owner competitiveness, land-use change and ES provision than climate-driven changes in land productivity. Other studies have also suggested that future socio-economic conditions are more important than climate change for land-based sectors (Brown et al. 2015; Brown et al. 2016c; Harrison et al. 2015; Harrison et al. 2016). This being the case, if ES demands are to be met by land-based sectors, mechanisms aimed at regulating societal demands will need to be implemented along with climate change mitigation strategies. In meeting demands for ES, demand regulation could be more effective than taking advantage from potentially increasing land productivities in Sweden. Strategies to affect global population growth (e.g. empowering women, guaranteeing education, providing access to safe and effective contraception) or consumption patterns have been tested in the past and can contribute to affect ES demands at the source (Chelleri et al. 2016; Spaargaren and Mol 2008; Worldwatch Institute 2012), although this is somewhat controversial. Alternatively, the development and introduction of new technologies may prove effective in increasing production so as to meet future societal demands. For instance, the use of genetic modification in producing tree varieties that are better adapted to future conditions, assisted gene flow, and assisted migration of existing tree species, show great potential in their capacity to increase the resilience and adaptability of forests and enhance the production of different ES (Fady et al. 2016; Johnston and Hesseln 2012; Plomion et al. 2016). *(Chapters 3 and 4)*

Land owner objectives and behaviour proved to substantially determine ES provision and the suitability of management strategies for adaptation. That owner objectives affect the provision of ES has often been suggested (Arano and Munn 2006; Sorice et al. 2014; Urquhart and Courtney 2011). However, this study is the first to assess how they affect the suitability of management strategies for adaptation in the forestry sector, in accounting for the effects of socio-economic and climatic change. Studies exploring adaptation strategies that account for farmer behaviour beyond profit maximisation have been performed for the agricultural sector (Acosta-Michlik and Espaldon 2008; Berger and Troost 2014; Brown et al. 2016a; Rebaudo and Dangles 2015; Wossen and Berger 2015). These studies showed how incorporating the effect of non-economic factors on decision-making and behaviour can result in a more accurate representation of adaptation choices made within a land-use modelling context, in turn improving understanding of how decision-making processes and behaviour impact on landscape and land-use change. This shows the need to account for owner behaviour in assessments of adaptation to global change and of land-use change, and in modelling assessments in particular, across land-based sectors. *(Chapters 3 and 4)*

Some key messages can be distilled from the scenario analysis. First, if the road towards a lower emission and more sustainable world is to be taken, forest expansion and an increase in the proportion of old growth forests (e.g. through extended rotation periods or nemoral broadleaf forests) can play a major role in Sweden towards its achievement. **Forest owners with sets of objectives that transcend profit making will likely be more competitive and have a greater presence under such a future scenario**. Different approaches are currently being used and promoted to achieve higher sustainability and lower emissions with forestry. Retention of trees at final felling is among the most common methods used in Scandinavian forests, and it continues to receive support as an effective means of preserving biodiversity in commercial forests (e.g. Heikkala et al. 2014; Lamas et al. 2015; Venier et al. 2015; Work et al. 2010). Monkkonen et al. (2009) suggested voluntary conservation programmes, land purchase for conservation, and landscape level planning in productive forest lands, in addition to protected areas, to maximise the production of biodiversity services in Fennoscandian forests. Others (Koskela et al. 2007; Roberge et al. 2016) have promoted extended rotations to achieve high provision levels of biodiversity, recreation and carbon sequestration. On the other hand, under a more carbon-intensive world where sustainability only plays a secondary or minor role, agriculture may be attributed greater importance when considering land-use priorities. **A large expansion in intensive agriculture can be expected especially under a carbon-fuelled technology-driven world**. To feed a rapidly growing population, sustainable intensification that relies on low emission, low environmental impact, and high spatially and temporally efficient methods of growing food have been suggested (Wu et al. 2014). These involve a proper allocation of crops in space and time, increasing the yield per unit area of individual crops, increasing the number of crops sown on a particular area of land, and the protection of fertile cropland (e.g. banning of conversion of high quality cropland to non-agricultural use). Also, knowledge transfer to farmers and to those working closely with them (advisors, rural practitioners) is essential to enable effective adaptive actions (Iglesias et al. 2012). Planned adaptation that accounts for competing pressures, drivers, and natural hazards is also necessary. It should also be noted that whichever scenario Sweden eventually follows will largely depend on developments elsewhere in the world. *(Chapters 3 and 4)*

Management strategies can be differently competitive under different future scenarios, and depending on local environmental conditions, socio-economic conditions at different scales (i.e. local to global), owner objectives and behaviour, and other contextual factors. It is therefore necessary that, in deciding for tree species to plant and management strategies to implement, forest owners and advisors consider their suitability for the particular location and under possible future changes in the determinants of suitability. Such an assessment will likely yield less uncertain results for lower-end climate change scenarios. That many of my findings were more uncertain under high-end climate change scenarios, implies that more extreme climatic futures will likely be more difficult to adapt to. Assessments of coping ability under a spread of future scenarios can be helpful in finding suitable adaptation management strategies. A strategy with a high coping ability is a strategy that, under most or all plausible futures, will be able to be at least as competitive as under current conditions. Such a strategy is therefore a good choice, even if future socio-environmental change is uncertain, if it proves to be competitive under current conditions. Thus, strategies scoring high in both aspects should be promoted. *(Chapter 4)*

1.2 Assessing and modelling adaptation

A famous quote concerning models states that: “All models are wrong, but some are useful” (Box and Draper 1987). Under this premise, I discuss in the following how ABMs can help to assess adaptation to global change in forestry and land-use change. The study limitations and how model representations of SES processes could be improved are discussed in the next section.

CRAFTY-Sweden and the institutional model developed here introduce novel concepts and approaches to the assessment of adaptation to global change in forestry. But also, the modelling results contributed important insights into the approaches, assumptions and drivers that should be considered in future modelling assessments of adaptation in socio-ecological systems. Overall, the findings suggest that model applications of this kind can be very useful in assessments of land-use change, ES provision, and adaptation to global change.

Due to the importance of ES demands in understanding the conditions and changes needed to fulfil societal needs for ES, I incorporated ES demands into the evaluation of benefits from ES provision. This is a novel approach in the assessments of adaptation and land-use change in the forestry sector. The approach proved insightful in allowing an assessment of the degree to which ES provision under different scenarios may be able to meet societal demands, and give recommendations about necessary measures to close the supply-demand gap.

The suitability of management strategies for adaptation is not a static, inherent characteristic of a system, but evolves in response to changing contexts that include both the external global change drivers and, importantly, the internal dynamics of agent interactions. This represents a change in paradigm in the way we assess the capacities of individuals and societies to adapt. It suggests that process-based models are more appropriate for the study of autonomous adaptation and future adaptive and coping capacity than models that assess these capacities using indicators based on discrete time snapshots or exogenous proxies, without accounting for interactions between agents that arise from their decisions. Other studies have also shown that agent interactions and the nature of those interactions can determine the success or failure (or the adaptive capacity) in adapting to a changing socio-environmental context. Processes of social learning, cooperation, self-organisation and innovation have proven critical in enabling successful adaptation (Boyd et al. 2011; Chhetri et al. 2012; Clark and Crabtree 2015; Marshall et al. 2009; Mathew and Perreault 2015; Pacheco et al. 2014). Hence, models capable of simulating agent behaviour and interactions in SES, such as ABMs or individual-based models (Railsback 2001), are best placed to take the lead in assessments aimed at informing adaptation. *(Chapter 4)*

Institutional conceptual models as presented here can support better understanding of the key institutional decision-making dynamics and their consequences, endogenously and flexibly across different SES. These models have a role in allowing scientists, policymakers and planners to better understand the consequences of institutional actions and interactions on SES. While, as with many models, institutional models face the challenge of reliable parameterisation when developing real world applications, data collection methods exist that could support this (e.g. social surveys and focus groups). Institutional models can be particularly useful in studying SES issues, such as land-use and land-cover change, natural resource management, or adaptation, when coupled with other SES models that simulate interactions between land managers and the environment (e.g. CRAFTY-Sweden). Coupling such models would be instrumental in exploring the feedbacks between institutions, land managers and the environment, and their consequences. *(Chapter 5)*

1.3 Limitations and future research

ABM was chosen because it allows to simulate the diversity in human behaviour within a population, and observe the consequences of individual decisions and their interactions. One of the main challenges posed by these models is the difficulty of their parameterisation, i.e., to find appropriate numeric values to represent the behavioural characteristics of individuals. Empirical data that is adequate to represent these behaviours is often non-existent or overly costly to obtain, hence limiting their application. Furthermore, the validation of ABMs (and especially their emergent outcomes) is at some level impossible, because social systems are never closed or static, and causality can never be attributed to quantitative social characteristics in isolation (Brown et al. 2016b). Nevertheless, we overcame the parameterisation and (to some extent) validation challenges by combining information gathered through a questionnaire survey, a literature review, and publicly available data from the Swedish Forest Agency. SFA and survey data were used to distribute forest owner types geographically, while survey data allowed the validation of the forest owner typology, conferring some robustness to our approach. However, even though some biophysical results could be validated (e.g. timber supply curve, being largely driven by past planting events, was validated using national forest age distributions from an alternative data source), outcomes that were largely determined by agent behaviour could not be validated (e.g. competition process) due to lack of data.

Translating real world processes into models also has its limitations, as models are simplifications of reality and can therefore only represent real world processes and phenomena to a limited extent. The accurate representation of these processes is largely dependent on the modeller's knowledge about the real world system, his/her/their capacity to appropriately represent these through algorithms, and the availability and suitability of data. Even when all these conditions apply, the purpose of the model will define the degree of complexity that needs to be represented. However, as the purpose of system modelling is to improve our understanding about systems, the fact that the system is not fully understood (e.g. what are the main drivers of system change) can limit in turn ones capacity to develop an accurate model at the desired scale and degree of complexity. Therefore, modelling is an iterative process through which the modeller can learn and continually adapt its models to learn about the real world system. This process can be useful and enlightening even when the accurate and complete representation of real world processes may never be achieved.

Studying the future with SES models carries substantial uncertainties that are associated to model complexity and the countless interactions that can occur between the model components. If we consider real world elements not represented as model components, the uncertainty inherent in model projections is even higher. This is especially so in models that represent the behaviour and/or decision-making of humans, like ABMs. The fact that humans often make irrational and inconsistent decisions makes human systems difficult to predict. In fact, if anything, given the multiple paths that human decisions and their interaction could potentially lead to in a SES, a well performing ABM should be expected to output a rather wide spectrum of highly variable and unpredictable emergent outcomes reflecting high uncertainty. This means that ABMs (as every other SES model) should not be used for prediction. Nevertheless, these models remain highly valuable heuristic and exploratory tools that can substantially improve our understanding of SES and the interactions taking place within them.

Future studies of adaptation and land-use change in SES will benefit from the incorporation of ES demands into the evaluation of benefits from ES provision. Nevertheless, CRAFTY-Sweden only incorporated a limited number of ES currently being discussed and valued in Sweden nationally, and similar models could benefit from including a more comprehensive set of ES. Water conservation and purification, flood and storm protection, air pollution absorption or cultural values are examples of ES provided by forests that were not considered here (Hansen and Malmaeus 2016; Ninan and Inoue 2013). Because different forest types and management strategies may contribute different levels of these services, their incorporation could alter the suitability of forest types and management strategies for adaptation, leading to different land-use transitions. Furthermore, changes in demands for these services could further accentuate the differences between models incorporating different sets of services. Nevertheless, demand for, and benefits from, these services can be hard to determine.

I assumed that the degree of importance attributed to meeting service demands does not change in the course of 90 years. However, demand could change through time, differently for each service, as a result of changes in societal needs and values. To illustrate this point, the importance attributed to meeting demands for food in a least developed country with food insecurity is greater than in a developed country where access to food is adequate and not vulnerable. Similarly, under a socio-economic scenario such as SSP4, characterised by social inequality, if we assume increasing social inequality and poverty to happen at the national level over time, the importance attributed to meeting food demands would grow over time too. The representation of the evolution of benefits obtained from the provision of different services would therefore benefit from characterising the importance of meeting demands as a time-dependent variable, which can be parameterised as part of scenario development.

The ways in which owners interact, who they interact with and the degrees of trust with which they interact strongly affect how they deal with complexity and uncertainty in the land-use system (Bodin and Norberg 2005; Huff et al. 2015; Kittredge et al. 2013; Wang et al. 2015). Similar factors affect interactions between institutions (Lubell 2015; Lucas and Baxter 2012; Ostrom 2009). In CRAFTY-Sweden, interactions between agents were limited to competition between land owners. In the conceptual institutional model, however, interactions and actions occurring between institutions were mapped, and could be represented in the institutional action model. The representation of agent networks and interactions has proven to be a determinant of the adoption of particular management strategies by land owners (Clark and Crabtree 2015; Manson et al. 2016; Olabisi et al. 2015). Therefore, their inclusion in CRAFTY-Sweden and similar ABMs would represent an improvement in the representation of agent decision-making processes and the consequences of this for land-use change.

Results regarding forest expansion towards the mountains need to be taken with caution as, given the varying topography, the 50 by 50 km resolution of the climate data from which productivity trends were derived (using LPJ-GUESS) might be too coarse to simulate the correct trends at the 1 by 1 km resolution (Fridley 2009; Griffiths et al. 2009). Studies that consider the impacts of climate change on productivities may therefore benefit from higher resolution data on productivity trends for areas with a heterogeneous topography.

Suggestions for further research into adaptation in forestry, beyond the realm of modelling, can also be drawn from this study. First, even though short rotations are currently being considered for risk mitigation, before they are broadly promoted, there is a need to further study the extent to which measures aiming to enhance the production of biodiversity and recreation, and carbon sequestration (e.g. extended rotations, set-asides for conservation) would be able to compensate for the loss associated with shorter rotation times. Also, further research is necessary on the effects of socio-economic change versus those caused by climatic change. Finally, future research into the suitability of forest management strategies to adapt to global change should look into both their competitiveness and their ability to cope under a spread of futures.

1.4 Policy relevance

If future Swedish in-house and overseas greenhouse CO_2 emission are to be met, LULUCF may need to be accounted for. Short stand rotations will not contribute to the recovery of carbon sequestration in forests up to its full potential. Instead, uneven-aged forest management, extended rotations and set-aside for conservation can contribute to a higher and more stable carbon storage in forests.

It is expected that the protection of forests with a high nature value will likely contribute to increasing national levels of biodiversity and recreation being supplied. I have suggested voluntary set-asides on the basis of my findings. I have also observed that the implementation of multi-objective forestry, and the diversification of management strategies that consider the particular spatio-temporal context also offer suitable solutions to achieve sustainable forestry. Short rotations, however, which have been suggested as a means of reducing risks from disturbances and contribute to climate mitigation, was found to be a detrimental solution if applied at a large scale. Extended rotation times, instead, may entail a higher risk in the face of disturbance, yet they can contribute to increase the provision of biodiversity, as well as recreation. Where extended rotations are implemented, other risk mitigating measures (e.g. planting trees with deep and dense root systems to prevent wind throw) can be included to compensate for the higher risk from longer rotation times. Retention of trees at final felling is commonly done today in Sweden as a way of preserving biodiversity in commercial forests. The implementation of uneven-aged forest management, especially if done at large scales, can have a crucial role in maintaining a constant provision of biodiversity and other ES through time. Lastly, global ES demand regulation through strategies to affect population growth and consumption patterns, and research into, and implementation of, new technologies to increase ES production can also play a major role in meeting future ES service demands.

References

Acosta-Michlik L, Espaldon V (2008) Assessing vulnerability of selected farming communities in the Philippines based on a behavioural model of agent's adaptation to global environmental change Global Environmental Change-Human and Policy Dimensions 18:554-563 doi:10.1016/j.gloenvcha.2008.08.006

Acosta-Michlik L, Klein RJT, Reidsma P, Metzger MJ, Rounsevell MDA, Leemans R, Schroter D (2013) A spatially explicit scenario-driven model of adaptive capacity to global change in Europe Global Environmental Change-Human and Policy Dimensions 23:1211-1224 doi:10.1016/j.gloenvcha.2013.03.008

Agrawal A, Yadama GN (1997) How do local institutions mediate market and population pressures on resources? Forest Panchayats in Kumaon, India Dev Change 28:435-465 doi:10.1111/1467-7660.00050

Alexandratos N, Bruinsma J (2012) World agriculture towards 2030/2050: The 2012 revision. FAO, Rome

Amacher GS, Conway CM, Sullivan J (2003) Econometric analyses of nonindustrial forest landowners: Is there anything left to study? Journal of Forest Economics 9:137-164 doi:http://dx.doi.org/10.1078/1104-6899-00028

Andersson M (2012) Assessing non-industrial private forest owners' attitudes to risk: Do owner and property characteristics matter? Journal of Forest Economics 18:3-13 doi:10.1016/j.jfe.2011.05.001

Andersson M, Gong PC (2010) Risk preferences, risk perceptions and timber harvest decisions - An empirical study of nonindustrial private forest owners in northern Sweden Forest Policy Econ 12:330-339 doi:10.1016/j.forpol.2010.02.002

Angelstam P, Andersson K, Axelsson R, Elbakidze M, Jonsson BG, Roberge JM (2011) Protecting forest areas for biodiversity in Sweden 1991-2010: the policy implementation process and outcomes on the ground Silva Fennica 45:1111-1133

Appelstrand M (2012) Developments in Swedish forest policy and administration - from a "policy of restriction" toward a "policy of cooperation" Scandinavian Journal of Forest Research 27:186-199 doi:10.1080/02827581.2011.635069

Arano KG, Munn IA (2006) Evaluating forest management intensity: A comparison among major forest landowner types Forest Policy Econ 9:237-248 doi:10.1016/j.forpol.2005.07.011

Araya-Muñoz D, Metzger MJ, Stuart N, Wilson M, Alvarez L (2016) Assessing urban adaptive capacity to climate change J Environ Manage 183:314-324 doi:10.1016/j.jenvman.2016.08.060

Arneth A, Brown C, Rounsevell MDA (2014) Global models of human decision-making for land-based mitigation and adaptation assessment Nature Clim Change 4:550-557 doi:10.1038/nclimate2250

Arnold JEM, Perez MR (2001) Can non-timber forest products match tropical forest conservation and development objectives? Ecol Econ 39:437-447 doi:10.1016/s0921-8009(01)00236-1

Baker AM, Lacy RC, Leus K, Traylor-Holzer K (2011) Intensive management of populations for conservation WAZA magazine 12:40-43

Batterbury SPJ, Bebbington AJ (1999) Environmental histories, access to resources and landscape change: An introduction Land Degrad Dev 10:279-289 doi:10.1002/(sici)1099-145x(199907/08)10:4<279::aid-ldr364>3.0.co;2-7

Beach RH, Pattanayak SK, Yang JC, Murray BC, Abt RC (2005) Econometric studies of non-industrial private forest management a review and synthesis Forest Policy Econ 7:261-281 doi:10.1016/s1389-9341(04)00065-0

Bellard C, Bertelsmeier C, Leadley P, Thuiller W, Courchamp F (2012) Impacts of climate change on the future of biodiversity Ecology Letters 15:365-377 doi:10.1111/j.1461-0248.2011.01736.x

Beratan KK (2007) A cognition-based view of decision processes in complex social-ecological systems Ecol Soc 12 doi:27

Berg A, Ehnstrom B, Gustafsson L, Hallingback T, Jonsell M, Weslien J (1994) Threatened plant, animal and fungus species in Swedish forests - Distribution and habitat associations Conserv Biol 8:718-731 doi:10.1046/j.1523-1739.1994.08030718.x

Berger T, Troost C (2014) Agent-based modelling of climate adaptation and mitigation options in agriculture Journal of Agricultural Economics 65:323-348 doi:10.1111/1477-9552.12045

Bjärstig T, Keskitalo ECH (2013) How to influence forest-related issues in the European Union? Preferred strategies among Swedish forest industry Forests 4:693-709 doi:10.3390/f4030693

Blennow K (2008) Risk management in Swedish forestry - Policy formation and fulfilment of goals Journal of Risk Research 11:237-254 doi:10.1080/13669870801939415

Blennow K (2012) Adaptation of forest management to climate change among private individual forest owners in Sweden Forest Policy Econ 24:41-47 doi:10.1016/j.forpol.2011.04.005

Blennow K, Andersson M, Sallnas O, Olofsson E (2010) Climate change and the probability of wind damage in two Swedish forests Forest Ecology and Management 259:818-830 doi:10.1016/j.foreco.2009.07.004

Blennow K, Persson J, Tome M, Hanewinkel M (2012) Climate change: believing and beeing implies adapting Plos One 7 doi:10.1371/journal.pone.0050182

Blennow K, Persson J, Wallin A, Vareman N, Persson E (2014) Understanding risk in forest ecosystem services: implications for effective risk management, communication and planning Forestry 87:219-228

Bodin O, Norberg J (2005) Information network topologies for enhanced local adaptive management Environmental Management 35:175-193 doi:10.1007/s00267-004-0036-7

Bohensky E (2014) Learning Dilemmas in a Social-Ecological System: An Agent-Based Modeling Exploration Jasss 17 doi:2

Boon TE, Broch SW, Meilby H (2010) How financial compensation changes forest owners' willingness to set aside productive forest areas for nature conservation in Denmark Scandinavian Journal of Forest Research 25:564-573 doi:10.1080/02827581.2010.512875

Boon TE, Meilby H, Thorsen BJ (2004) An empirically based typology of private forest owners in Denmark: Improving communication between authorities and owners Scandinavian Journal of Forest Research 19:45-55 doi:10.1080/14004080410034056

Boone RB, Galvin KA, BurnSilver SB, Thornton PK, Ojima DS, Jawson JR (2011) Using Coupled Simulation Models to Link Pastoral Decision Making and Ecosystem Services Ecol Soc 16

Bouma J, Varallyay G, Batjes N (1998) Principal land use changes anticipated in Europe Agriculture, Ecosystems & Environment 67:103-119

Boumans R et al. (2002) Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model Ecol Econ 41:529-560 doi:10.1016/s0921-8009(02)00098-8

Box GEP, Draper NR (1987) Empirical model-building and response surfaces. John Wiley & Sons, New York, NY

Boyd R, Richerson PJ, Henrich J (2011) The cultural niche: Why social learning is essential for human adaptation Proc Natl Acad Sci U S A 108:10918-10925 doi:10.1073/pnas.1100290108

Bravo G (2014) Empirical agent-based modelling - Challenges and solutions: Volume 1, the characterisation and parameterisation of empirical agent-based models Jasss 17

Briegel R, Ernst A, Holzhauer S, Klemm D, Krebs F, Martínez Piñánez A Social-ecological modelling with LARA: A psychologically well-founded lightweight agent architecture. In: Seppelt R, Voinov AA, Lange S, Bankamp D (eds) International Environmental Modelling and Software Society (iEMSs) 2012 International Congress on Environmental Modelling and Software. Managing Resources of a Limited Planet: Pathways and Visions under Uncertainty, Sixth Biennial Meeting, Leipzig, Germany, 2012.

Brown C, Bakam I, Smith P, Matthews R (2016a) An agent-based modelling approach to evaluate factors influencing bioenergy crop adoption in north-east Scotland Global Change Biology Bioenergy 8:226-244 doi:10.1111/gcbb.12261

Brown C, Brown E, Murray-Rust D, Cojocaru G, Savin C, Rounsevell M (2015) Analysing uncertainties in climate change impact assessment across sectors and scenarios Clim Change 128:293-306 doi:10.1007/s10584-014-1133-0

Brown C, Brown K, Rounsevell M (2016b) A philosophical case for process-based modelling of land use change Modeling Earth Systems and Environment 2:1-12 doi:10.1007/s40808-016-0102-1

Brown C, Holzhauer S, Metzger MJ, Paterson JS, Rounsevell M (2016c) Land managers' behaviours modulate pathways to visions of future land systems Regional Environmental Change:1-15 doi:10.1007/s10113-016-0999-y

Brown C, Murray-Rust D, van Vliet J, Alam SJ, Verburg PH, Rounsevell MD (2014) Experiments in globalisation, food security and land use decision making Plos One 9 doi:10.1371/journal.pone.0114213

Bryson J, Piper J, Rounsevell M (2010) Envisioning futures for climate change policy development: Scenarios use in European environmental policy institutions Environ Policy Gov 20:283-294 doi:10.1002/eet.542

Buonocore E, Franzese PP, Ulgiati S (2012) Assessing the environmental performance and sustainability of bioenergy production in Sweden: A life cycle assessment perspective Energy 37:69-78 doi:10.1016/j.energy.2011.07.032

Campo PC, Mendoza GA, Guizol P, Villanueva TR, Bousquete F (2009) Exploring management strategies for community-based forests using multi-agent systems: A case study in Palawan, Philippines J Environ Manage 90:3607-3615 doi:DOI 10.1016/j.jenvman.2009.06.016

Canadas MJ, Novais A (2014) Bringing local socioeconomic context to the analysis of forest owners' management Land Use Policy 41:397-407 doi:http://dx.doi.org/10.1016/j.landusepol.2014.06.017

Carter TR, Rounsevell MDA, Fronzek S, Harrison PA, Holman IP, Pirttioja NK (2015) Integrated assessment approach. EU FP7 IMPRESSIONS Project Deliverable D3.1

Carter TR, Saarikko RA, Niemi KJ (1996) Assessing the risks and uncertainties of regional crop potential under a changing climate in Finland Agricultural and Food Science in Finland 5:329-350

Carvalho-Ribeiro SM, Lovett A, O'Riordan T (2010) Multifunctional forest management in Northern Portugal: Moving from scenarios to governance for sustainable development Land Use Policy 27:1111-1122 doi:10.1016/j.landusepol.2010.02.008

Cashore B, Auld AG, Newsom D (2004) Governing Through Markets - Forest Certification and the Emergence of Non-state. Yale University Press, New Haven, CT

Caughley G (1994) Directions in Conservation Biology Journal of Animal Ecology 63:215-244 doi:10.2307/5542

Chelleri L, Kua HW, Sanchez JPR, Nahiduzzaman KM, Thondhlana G (2016) Are people responsive to a more sustainable, decentralized, and user-driven management of urban metabolism? Sustainability 8 doi:10.3390/su8030275

Chhetri N, Chaudhary P, Tiwari PR, Yadaw RB (2012) Institutional and technological innovation: Understanding agricultural adaptation to climate change in Nepal Applied Geography 33:142-150 doi:10.1016/j.apgeog.2011.10.006

Cienciala E, Cerny M, Apltauer J, Exnerova Z (2005) Biomass functions applicable to European beech Journal of Forest Science (Prague) 51:147-154

Clark JK, Crabtree SA (2015) Examining social adaptations in a volatile landscape in northern Mongolia via the agent-based model Ger Grouper Land 4:157-181 doi:10.3390/land4010157

Cocklin C, Dibden J, Mautner N (2006) From market to multifunctionality? Land stewardship in Australia Geogr J 172:197-205 doi:10.1111/j.1475-4959.2006.00206.x

Conway MC, Amacher GS, Sullivan J, Wear D (2003) Decisions nonindustrial forest landowners make: an empirical examination Journal of Forest Economics 9:181-203 doi:http://dx.doi.org/10.1078/1104-6899-00034

Costanza R et al. (1997) The value of the world's ecosystem services and natural capital Nature 387:253-260 doi:10.1038/387253a0

Creighton J, Baumgartner D, Blatner K (2002) Ecosystem management and nonindustrial private forest landowners in Washington State, USA Small-scale Forest Economics, Management and Policy 1:55-69 doi:10.1007/s11842-002-0005-z

Dawson TP, Jackson ST, House JI, Prentice IC, Mace GM (2011) Beyond predictions: Biodiversity conservation in a changing climate Science 332:53-58 doi:10.1126/science.1200303

De Groot R et al. (2010) Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: Kumar P (ed) The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, London, Washington, pp 9-40

Dearing JA, Braimoh AK, Reenberg A, Turner BL, van der Leeuw S (2010) Complex Land Systems: the Need for Long Time Perspectives to Assess their Future Ecol Soc 15 doi:21

Douglas P (1976) The Cobb-Douglas production function once again: Its history, its testing, and some new empirical values Journal of Political Economy 84:903–915 doi:10.1086/260489

Dovers SR, Hezri AA (2010) Institutions and policy processes: the means to the ends of adaptation Wiley Interdisciplinary Reviews-Climate Change 1:212-231 doi:10.1002/wcc.29

Duncker PS, Barreiro SM, Hengeveld GM, Lind T, Mason WL, Ambrozy S, Spiecker H (2012a) Classification of forest management approaches: a new conceptual framework and its applicability to European forestry Ecol Soc 17 doi:10.5751/es-05262-170451

Duncker PS et al. (2012b) How forest management affects ecosystem services, including timber production and economic return: synergies and trade-Offs Ecol Soc 17 doi:10.5751/es-05066-170450

Eagly AH, Chaiken S (1993) The psychology of attitudes. Harcourt Brace Jovanovich, Fort Worth, USA

Easterling WE et al. (2007) Food, fibre and forest products. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.

Edwards DM et al. (2012) Public preferences across Europe for different forest stand types as sites for recreation Ecol Soc 17 doi:10.5751/es-04520-170127

EEA (2014) Raster data on land cover for the CLC2006 inventory - Version 17. http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2006-raster-3#tab-gis-data. Accessed 04 March 2015

EEA (2015) Nationally designated areas (CDDA). http://www.eea.europa.eu/data-and-maps/data/nationally-designated-areas-national-cdda-10#tab-european-data. Accessed 04 March 2015

Eggers J, Lamas T, Lind T, Ohman K (2014) Factors Influencing the Choice of Management Strategy among Small-Scale Private Forest Owners in Sweden Forests 5:1695-1716 doi:10.3390/f5071695

Eko PM (1985) En produktionsmodell for skog i Sverige, baserad pa bestand fran riksskogstaxeringens provytor [A growth simulator for Swedish forests, based on data from the National Forest Survey]. Swedish University of Agricultural Sciences

Elkington J (1994) Towards the sustainable corporation - win-win-win business strategies for sustainable development Calif Manage Rev 36:90-100

Emtage N, Herbohn J, Harrison S (2007) Landholder profiling and typologies for natural resource-management policy and program support: Potential and constraints Environmental Management 40:481-492 doi:10.1007/s00267-005-0359-z

Engström K et al. (2016) Assessing uncertainties in global cropland futures using a conditional probabilistic modelling framework Earth Syst Dynam doi:10.5194/esd-2016-7

Eriksson L (2012) Exploring underpinnings of forest conflicts: a study of forest values and beliefs in the general public and among private forest owners in Sweden Society & Natural Resources 25:1102-1117 doi:10.1080/08941920.2012.657749

European Commission (2014) EU forests and forest related policies. http://ec.europa.eu/environment/forests/home_en.htm. Accessed 22 August 2014

Fady B, Cottrell J, Ackzell L, Alia R, Muys B, Prada A, Gonzalez-Martinez SC (2016) Forests and global change: what can genetics contribute to the major forest management and policy challenges of the twenty-first century? Regional Environmental Change 16:927-939 doi:10.1007/s10113-015-0843-9

FAO (2000) The global outlook for future wood supply from forest plantations. FAO, Rome

FAO (2005) Trends in wood products 1961-2003. FAO, Rome

FAO (2012) Global forest products products facts and figures. http://www.fao.org/forestry/en/. Accessed 28/03/2014

FAO (2015) FAO Statistics Division - Production. http://faostat3.fao.org/browse/Q/*/E. Accessed 15 December 2015

Felton A et al. (2016) How climate change adaptation and mitigation strategies can threaten or enhance the biodiversity of production forests: Insights from Sweden Biological Conservation 194:11-20 doi:10.1016/j.biocon.2015.11.030

Ferber J (1999) Multi-agent systems: an introduction to distributed artifitial intelligence. Addison-Wesley Longman, Harlow, UK

Ficko A, Boncina A (2013) Probabilistic typology of management decision making in private forest properties Forest Policy Econ 27:34-43 doi:10.1016/j.forpol.2012.11.001

Finley AO, Kittredge DB (2006) Thoreau, Muir, and Jane Doe: Different types of private forest owners need different kinds of forest management North J Appl For 23:27-34

Foley JA et al. (2005) Global consequences of land use Science 309:570-574 doi:10.1126/science.1111772

Fontaine CM, Rounsevell MDA, Barbette AC (2014) Locating household profiles in a polycentric region to refine the inputs to an agent-based model of residential mobility Environment and Planning B-Planning & Design 41:163-184 doi:10.1068/b37072

Franklin JF (1993) Preserving biodiversity: species, ecosystems, or landscapes? Ecological Applications 3:202-205 doi:10.2307/1941820

Fridley JD (2009) Downscaling climate over complex terrain: high finescale (< 1000 m) spatial variation of near-ground temperatures in a montane forested landscape (Great Smoky Mountains) Journal of Applied Meteorology and Climatology 48:1033-1049 doi:10.1175/2008jamc2084.1

Fujimori T (2001) Ecological and silvicultural strategies for sustainable forest management. Elsevier, Amsterdam

Fulginiti LE, Perrin RK (1998) Agricultural productivity in developing countries Agricultural Economics 19:45-51 doi:10.1016/s0169-5150(98)00045-0

Gamfeldt L et al. (2013) Higher levels of multiple ecosystem services are found in forests with more tree species Nat Commun 4 doi:10.1038/ncomms2328

Gauthier S et al. (2014) Climate change vulnerability and adaptation in the managed Canadian boreal forest Environmental Reviews 22:256-285 doi:10.1139/er-2013-0064

Godfray HCJ et al. (2010) Food security: The challenge of feeding 9 billion people Science 327:812-818 doi:10.1126/science.1185383

Griffiths RP, Madritch MD, Swanson AK (2009) The effects of topography on forest soil characteristics in the Oregon Cascade Mountains (USA): Implications for the effects of climate change on soil properties Forest Ecology and Management 257:1-7 doi:10.1016/j.foreco.2008.08.010

Griliches Z, Mason WM (1972) Education, Income, and Ability Journal of Political Economy 80:S74-S103 doi:10.2307/1831252

Grimm V et al. (2006) A standard protocol for describing individual-based and agent-based models Ecol Model 198:115-126 doi:http://dx.doi.org/10.1016/j.ecolmodel.2006.04.023

Grimm V, Berger U, DeAngelis DL, Polhill JG, Giske J, Railsback SF (2010) The ODD protocol: A review and first update Ecol Model 221:2760-2768 doi:http://dx.doi.org/10.1016/j.ecolmodel.2010.08.019

Groot JCJ et al. (2009) On the contribution of modelling to multifunctional agriculture: Learning from comparisons J Environ Manage 90:S147-S160 doi:10.1016/j.jenvman.2008.11.030

Guillem EE, Barnes AP, Rounsevell MDA, Renwick A (2012) Refining perception-based farmer typologies with the analysis of past census data J Environ Manage 110:226-235 doi:10.1016/j.jenvman.2012.06.020

Guzy MR, Smith CL, Bolte JP, Hulse DW, Gregory SV (2008) Policy Research Using Agent-Based Modeling to Assess Future Impacts of Urban Expansion into Farmlands and Forests Ecol Soc 13 doi:37

Hagberg E, Matérn B (1975) Tabeller för kubering av ek och bok vol Rapporter och Uppsatser 14. Skogshögskolan, Inst. f. skoglig matematisk statistik, Stockholm

Hägglund B, Lundmark JE (1987) Handledning I Bonitering med Skogshögsskolans boniteringssystem, Del 2 Diagram och tabeller. Skogsstyrelsen, Jönköping

Haines-Young R, Potschin M (2013) Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012. EEA Framework Contract No. EEA/IEA/09/003

Hamilton SH, ElSawah S, Guillaume JHA, Jakeman AJ, Pierce SA (2015) Integrated assessment and modelling: Overview and synthesis of salient dimensions Environ Modell Softw 64:215-229 doi:10.1016/j.envsoft.2014.12.005

Hanewinkel M, Cullmann DA, Schelhaas MJ, Nabuurs GJ, Zimmermann NE (2013) Climate change may cause severe loss in the economic value of European forest land Nature Climate Change 3:203-207 doi:10.1038/nclimate1687

Hannah L, Costello C, Guo C, Ries L, Kolstad C, Panitz D, Snider N (2011) The impact of climate change on California timberlands Clim Change 109:429-443 doi:10.1007/s10584-011-0307-2

Hansen K, Malmaeus M (2016) Ecosystem services in Swedish forests Scandinavian Journal of Forest Research:1-15 doi:10.1080/02827581.2016.1164888

Haque MS (1999) The fate of sustainable development under neo-liberal regimes in developing countries Int Polit Sci Rev 20:197-218 doi:10.1177/0192512199202005

Hare M, Deadman P (2004) Further towards a taxonomy of agent-based simulation models in environmental management Math Comput Simul 64:25-40 doi:10.1016/s0378-4754(03)00118-6

Harfoot M, Tittensor DP, Newbold T, McInerny G, Smith MJ, Scharlemann JPW (2014) Integrated assessment models for ecologists: the present and the future Global Ecology and Biogeography 23:124-143 doi:10.1111/geb.12100

Harrison PA, Dunford R, Savin C, Rounsevell MDA, Holman IP, Kebede AS, Stuch B (2015) Cross-sectoral impacts of climate change and socio-economic change for multiple, European land- and water-based sectors Clim Change 128:279-292 doi:10.1007/s10584-014-1239-4

Harrison PA, Dunford RW, Holman IP, Rounsevell MDA (2016) Climate change impact modelling needs to include cross-sectoral interactions Nature Climate Change:885–890 doi:10.1038/nclimate3039

Harrison PA, Holman IP, Cojocaru G, Kok K, Kontogianni A, Metzger MJ, Gramberger M (2013) Combining qualitative and quantitative understanding for exploring cross-sectoral climate change impacts, adaptation and vulnerability in Europe Regional Environmental Change 13:761-780 doi:10.1007/s10113-012-0361-y

Hayha T, Franzese PP, Paletto A, Fath BD (2015) Assessing, valuing, and mapping ecosystem services in Alpine forests Ecosystem Services 14:12-23 doi:10.1016/j.ecoser.2015.03.001

Heikkala O, Suominen M, Junninen K, Hamalainen A, Kouki J (2014) Effects of retention level and fire on retention tree dynamics in boreal forests Forest Ecology and Management 328:193-201 doi:10.1016/j.foreco.2014.05.022

Hogl K, Pregernig M, Weiss G (2005) What is new about new forest owners? A typology of private forest ownership in Austria Small-scale Forest Economics, Management and Policy 4:325-342 doi:10.1007/s11842-005-0020-y

Holm SO (2015) A management strategy for multiple ecosystem services in boreal forests Journal of Sustainable Forestry 34:358-379 doi:10.1080/10549811.2015.1009633

Huff ES, Leahy JE, Hiebeler D, Weiskittel AR, Noblet CL (2015) An agent-based model of private woodland owner management behavior using social interactions, information flow, and peer-to-peer networks Plos One 10 doi:10.1371/journal.pone.0142453

Hysing E (2009) Governing without government? The private governance of forest certification in Sweden Public Adm 87:312-326 doi:10.1111/j.1467-9299.2008.01750.x

Iglesias A, Quiroga S, Moneo M, Garrote L (2012) From climate change impacts to the development of adaptation strategies: Challenges for agriculture in Europe Clim Change 112:143-168 doi:10.1007/s10584-011-0344-x

IIASA (2015) SSP Database (Shared Socioeconomic Pathways) - Version 1.0. https://tntcat.iiasa.ac.at/SspDb/dsd?Action=htmlpage&page=about. Accessed 02 November 2015

Ingemarson F, Lindhagen A, Eriksson L (2006) A typology of small-scale private forest owners in Sweden Scandinavian Journal of Forest Research 21:249-259 doi:10.1080/02827580600662256

Institute of Environmental Studies (2015) Agricultural land use intensity data. VU University Amsterdam. http://www.falw.vu.nl/en/research/earth-sciences/earth-and-climate-cluster/research/environmental-geography/projects/agricultural-land-use-intensity-data.aspx. Accessed 05 September 2015

IPCC (2006) 2006 IPCC guidelines for national greenhouse gas inventories. IGES, Japan

IPCC (2007) Climate change 2007: Impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK

IPCC (2012) Managing the risks of extreme events and disasters to advance climate change adaptation. A Special Report of Working Groups I and II of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK, and New York, USA

IPCC (2014a) Climate change 2014: Impacts, adaptation and vulnerability. Part B: Regional aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Pannel on Climate Change. Cambridge, United Kingdom and New York, NY, USA

IPCC (2014b) Climate change 2014: Impacts, adaptation, and vulnerability. Part A: Global and sectoral aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change Cambridge, UK, and New York, USA

Jacob D et al. (2014) EURO-CORDEX: new high-resolution climate change projections for European impact research Regional Environmental Change 14:563-578 doi:10.1007/s10113-013-0499-2

Janse G, Ottitsch A (2005) Factors influencing the role of Non-Wood Forest Products and Services Forest Policy Econ 7:309-319 doi:10.1016/s1389-9341(03)00068-6

Janssen MA, Ostrom E (2006) Empirically based, agent-based models Ecol Soc 11 doi:37

Johansson U, Ekö PM, Elfving B, Johansson T, Nilsson U (2013) Fakta Skog - Nya höjdutvecklingskurvor för bonitering. Swedish University of Agricultural Sciences, Linköping

Johnston M, Hesseln H (2012) Climate change adaptive capacity of the Canadian forest sector Forest Policy Econ 24:29-34 doi:10.1016/j.forpol.2012.06.001

Jones C, Giorgi F, Asrar G (2011) CLIVAR Exchanges - The Coordinated Regional Downscaling Experiment: CORDEX, An international downscaling link to CMIP5 vol 16.

Jonsell M, Weslien J, Ehnstrom B (1998) Substrate requirements of red-listed saproxylic invertebrates in Sweden Biodiversity and Conservation 7:749-764 doi:10.1023/a:1008888319031

Jonsson AM, Lagergren F, Smith B (2015) Forest management facing climate change - an ecosystem model analysis of adaptation strategies Mitigation and Adaptation Strategies for Global Change 20:201-220 doi:10.1007/s11027-013-9487-6

Jonsson R, Egnell G, Baudin A (2011) Swedish forest sector outlook study - Geneva timber and forest discussion paper 58. United Nations, Geneva

Jordens Vänner (2014) Friends of the Earth Sweden - Jordens Vänner. http://www.jordensvanner.se/in-english. Accessed 21 August 2014

Joshi S, Arano KG (2009) Determinants of private forest management decisions: A study on West Virginia NIPF landowners Forest Policy Econ 11:118-125 doi:http://dx.doi.org/10.1016/j.forpol.2008.10.005

Karali E, Brunner B, Doherty R, Hersperger AM, Rounsevell MDA (2013) The effect of farmer attitudes and objectives on the heterogeneity of farm attributes and management in switzerland Hum Ecol 41:915-926 doi:10.1007/s10745-013-9612-x

Karppinen H (1998) Values and objectives of non-industrial private forest owners in Finland Silva Fennica 32:43-59

Kengen S (1997) Forest valuation for decision making - Lessons of experience and proposalsfor improvement. FAO, Rome

Keskitalo ECH (2011) How can forest management adapt to climate change? Possibilities in different forestry systems Forests 2:415-430 doi:10.3390/f2010415

Keskitalo ECH (2013) Understanding Adaptive Capacity in Forest Governance: Editorial Ecol Soc 18 doi:10.5751/es-05924-180445

Keskitalo ECH, Liljenfeldt J (2012) Working with sustainability: Experiences of sustainability processes in Swedish municipalities Nat Resour Forum 36:16-27 doi:10.1111/j.1477-8947.2012.01442.x

Keskitalo ECH, Pettersson M (2012) Implementing multi-level governance? The legal basis and implementation of the EU Water Framework Directive for forestry in Sweden Environ Policy Gov 22:90-103 doi:10.1002/eet.1574

Keskitalo ECH, Westerhoff L, Juhola S (2012) Agenda-setting on the environment: the development of climate change adaptation as an issue in European states Environ Policy Gov 22:381-394 doi:10.1002/eet.1579

Kilgore MA, Snyder SA, Schertz J, Taff SJ (2008) What does it take to get family forest owners to enroll in a forest stewardship-type program? Forest Policy Econ 10:507-514 doi:10.1016/j.forpol.2008.05.003

Kittredge DB, Rickenbach MG, Knoot TG, Snellings E, Erazo A (2013) It's the network: how personal connections shape decisions about private forest use North J Appl For 30:67-74 doi:10.5849/njaf.11-004

Kjaer ED, Lobo A, Myking T (2014) The role of exotic tree species in Nordic forestry Scandinavian Journal of Forest Research 29:323-332 doi:10.1080/02827581.2014.926098

Kleinschmit D, Edwards P (2013) Pan-European forest-focused and forest-related policies. In: Pülzl H et al. (eds) European Forest Governance: issues at stake and the way forward. European Forest Institute, Joensuu, pp 45-51

Kleinschmit D, Ingemarson F, Holmgren S (2012) Research on forest policy in Sweden - Review Scandinavian Journal of Forest Research 27:120-129 doi:10.1080/02827581.2011.635076

Kline JD, Alig RJ, Johnson RL (2000a) Forest owner incentives to protect riparian habitat Ecol Econ 33:29-43 doi:10.1016/s0921-8009(99)00116-0

Kline JD, Alig RJ, Johnson RL (2000b) Fostering the production of nontimber services among forest owners with heterogeneous objectives Forest Science 46:302-311

Kok K, Pedde S, Jäger J, Harrison P (2015) European Shared Socioeconomic Pathways vol 2015. EU FP7 IMPRESSIONS Report

Koskela E, Ollikainen M, Pukkala T (2007) Biodiversity conservation in commercial boreal forestry: The optimal rotation age and retention tree volume Forest Science 53:443-452

Kullman L, Kjallgren L (2006) Holocene pine tree-line evolution in the Swedish Scandes: Recent tree-line rise and climate change in a long-term perspective Boreas 35:159-168 doi:10.1080/03009480500359137

Kunskap Direkt (2015) Minimiålder. http://www.kunskapdirekt.se/sv/KunskapDirekt/Avverka/Grunder/Skogsvardslagen/Minimialder/. Accessed 03 August 2015

Kuuluvainen T, Tahvonen O, Aakala T (2012) Even-aged and uneven-aged forest management in boreal Fennoscandia: a review Ambio 41:720-737 doi:10.1007/s13280-012-0289-y

Kvarda ME (2004) 'Non-agricultural forest owners' in Austria – a new type of forest ownership Forest Policy Econ 6:459-467 doi:http://dx.doi.org/10.1016/j.forpol.2004.01.005

Lafond V, Lagarrigues G, Cordonnier T, Courbaud B (2014) Uneven-aged management options to promote forest resilience for climate change adaptation: effects of group selection and harvesting intensity Ann For Sci 71:173-186 doi:10.1007/s13595-013-0291-y

Lagergren F, Jonsson AM, Blennow K, Smith B (2012) Implementing storm damage in a dynamic vegetation model for regional applications in Sweden Ecol Model 247:71-82 doi:10.1016/j.ecolmodel.2012.08.011

Laiho O, Lahde E, Pukkala T (2011) Uneven- vs even-aged management in Finnish boreal forests Forestry 84:547-556 doi:10.1093/forestry/cpr032

Lamas T, Sandstrom E, Jonzen J, Olsson H, Gustafsson L (2015) Tree retention practices in boreal forests: what kind of future landscapes are we creating? Scandinavian Journal of Forest Research 30:526-537 doi:10.1080/02827581.2015.1028435

Lambin EF, Geist HJ, Lepers E (2003) Dynamics of land-use and land-cover change in tropical regions Annu Rev Environ Resour 28:205-241 doi:10.1146/annurev.energy.28.050302.105459

Landais E (1998) Modelling farm diversity new approaches to typology building in France Agric Syst 58:505-527 doi:10.1016/s0308-521x(98)00065-1

Lantbrukarnas Riksförbund (2014) Swedish family forestry. http://www.lrf.se/globalassets/dokument/om-lrf/in-english/lrf-forestry/swedishfamilyforestrymars2014l.pdf.

Larson C (2013) Climate change. Losing arable land, China faces stark choice: adapt or go hungry Science 339:644-645

Lavorel S et al. (2007) Plant functional types : are we getting closer to the Holy Grail? In: Canadell JG, Pataki DE, Pitelka LF (eds) Terrestrial ecosystems in a changing World. Springer Berlin Heidelberg, Berlin, Heidelberg, pp 149–164. doi:10.1007/978-3-540-32730-1_13

Le Du-Blayo L (2011) How Do We Accommodate New Land Uses in Traditional Landscapes? Remanence of Landscapes, Resilience of Areas, Resistance of People Landsc Res 36:417-434 doi:10.1080/01426397.2011.583010

Le Goff H, Leduc A, Bergeron Y, Flannigan M (2005) The adaptive capacity of forest management to changing fire regimes in the boreal forest of Quebec Forestry Chronicle 81:582-592

Lengnick-Hall CA, Beck TE (2005) Adaptive fit versus robust transformation: How organizations respond to environmental change J Manag 31:738-757 doi:10.1177/0149206305279367

Levina E, Tirpak D (2006) Adaptation to climate change: key terms. OECD/IEA, Paris, France

Liao XC, Zhang YQ (2008) An econometric analysis of softwood production in the US South: a comparison of industrial and nonindustrial forest ownerships For Prod J 58:69-74

Lidestav G (2010) In competition with a brother: Women's inheritance positions in contemporary Swedish family forestry Scandinavian Journal of Forest Research 25:14-24 doi:10.1080/02827581.2010.506781

Lin BB (2011) Resilience in agriculture through crop diversification: adaptive management for environmental change Bioscience 61:183-193 doi:10.1525/bio.2011.61.3.4

Lindlof TR, Taylor BC (2011) Qualitative Communication Research Methods. SAGE, Thousand Oaks, USA

Lindner M et al. (2010) Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems Forest Ecology and Management 259:698-709 doi:10.1016/j.foreco.2009.09.023

Lo AY (2013) Carbon trading in a socialist market economy: Can China make a difference? Ecol Econ 87:72-74 doi:http://dx.doi.org/10.1016/j.ecolecon.2012.12.023

Löf M, Møller-Madsen E, Rytter L (2009) Skötsel av ädellövskog. Skogsstyrelsen,

Lönnstedt L (2012) Small Scale Forest Owners' Responsibilities: Results from a Swedish Case Study Small-Scale Forestry 11:407-416 doi:10.1007/s11842-011-9187-6

Lubell M (2015) Collaborative partnerships in complex institutional systems Current Opinion in Environmental Sustainability 12:41-47 doi:10.1016/j.cosust.2014.08.011

Lucas JW, Baxter AR (2012) Power, influence, and diversity in organizations Ann Am Acad Polit Soc Sci 639:49-70 doi:10.1177/0002716211420231

Lundstrom J, Ohman K, Ronnqvist M, Gustafsson L (2014) How reserve selection is affected by preferences in Swedish boreal forests Forest Policy Econ 41:40-50 doi:10.1016/j.forpol.2013.12.007

Lung T, Lavalle C, Hiederer R, Dosio A, Bouwer LM (2013) A multi-hazard regional level impact assessment for Europe combining indicators of climatic and non-climatic change Global Environmental Change-Human and Policy Dimensions 23:522-536 doi:10.1016/j.gloenvcha.2012.11.009

Majumdar I, Teeter L, Butler B (2008) Characterizing Family Forest Owners: A Cluster Analysis Approach Forest Science 54:176-184

Manson SM, Jordan NR, Nelson KC, Brummel RF (2016) Modeling the effect of social networks on adoption of multifunctional agriculture Environ Modell Softw 75:388-401 doi:10.1016/j.envsoft.2014.09.015

Marchetti M (2004) Monitoring and indicators of forest biodiversity in Europe - From ideas to operationality. European Forest Institute, Saarijarvi, Finland

Marshall NA, Marshall PA, Tamelander J, Obura D, Malleret-King D, Cinner JE (2009) A framework for social adaptation to climate change; sustaining tropical coastal communities and industries. IUCN, Gland, Switzerland

Martin W, Mitra D (2001) Productivity growth and convergence in agriculture versus manufacturing Economic Development and Cultural Change 49:403-422 doi:10.1086/452509

Mathew S, Perreault C (2015) Behavioural variation in 172 small-scale societies indicates that social learning is the main mode of human adaptation Proceedings of the Royal Society B-Biological Sciences 282 doi:10.1098/rspb.2015.0061

Matilainen A, Lahdesmaki M (2014) Nature-based tourism in private forests: Stakeholder management balancing the interests of entrepreneurs and forest owners? J Rural Stud 35:70-79 doi:10.1016/j.jrurstud.2014.04.007

Matthews RB, Gilbert NG, Roach A, Polhill JG, Gotts NM (2007) Agent-based land-use models: a review of applications Landscape Ecol 22:1447-1459 doi:DOI 10.1007/s10980-007-9135-1

Mazziotta A, Trivino M, Tikkanen OP, Kouki J, Strandman H, Monkkonen M (2015) Applying a framework for landscape planning under climate change for the conservation of biodiversity in the Finnish boreal forest Glob Change Biol 21:637-651 doi:10.1111/gcb.12677

MCPFE (2003) Improved Pan-European indicators for sustainable forest management. MCPFE Liaison Unit Vienna, Vienna, Austria

Meidinger E (2006) The administrative law of global private-public regulation: The case of forestry Eur J Int Law 17:47-87 doi:10.1093/ejil/chi168

Metzger MJ, Rounsevell MDA, Acosta-Michlik L, Leemans R, Schrotere D (2006) The vulnerability of ecosystem services to land use change Agric Ecosyst Environ 114:69-85 doi:10.1016/j.agee.2005.11.025

Metzger MJ, Schroter D, Leemans R, Cramer W (2008) A spatially explicit and quantitative vulnerability assessment of ecosystem service change in Europe Regional Environmental Change 8:91-107 doi:10.1007/s10113-008-0044-x

Meyfroidt P, Rudel TK, Lambin EF (2010) Forest transitions, trade, and the global displacement of land use Proc Natl Acad Sci U S A 107:20917-20922 doi:10.1073/pnas.1014773107

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: Synthesis. Washington, DC

Minx J, Scott K, Peters G, Barrett J (2008) An analysis of Sweden's carbon footprint. WWF, Solna, Sweden

Monk J (2014) How long should we ignore imperfect detection of species in the marine environment when modelling their distribution? Fish Fish 15:352-358 doi:10.1111/faf.12039

Monkkonen M, Ylisirnio AL, Hamalainen T (2009) Ecological efficiency of voluntary conservation of boreal-forest biodiversity Conserv Biol 23:339-347 doi:10.1111/j.1523-1739.2008.01082.x

Morgan G, Campbell JL, Crouch C, Perdersen OK, Whitley R (2010) Oxford handbook of comparative institutional analysis. Oxford University Press, New York

Mosnier A et al. (2014) Global food markets, trade and the cost of climate change adaptation Food Secur 6:29-44 doi:10.1007/s12571-013-0319-z

Murray-Rust D, Brown C, van Vliet J, Alam SJ, Robinson DT, Verburg PH, Rounsevell M (2014) Combining agent functional types, capitals and services to model land use dynamics Environ Modell Softw 59:187-201 doi:http://dx.doi.org/10.1016/j.envsoft.2014.05.019

Murray-Rust D, Dendoncker N, Dawson TP, Acosta-Michlik L, Karali E, Rounsevell M (2011) Conceptualising the analysis of socio-ecological systems through ecosystem services and agent-based modelling Journal of Land Use Science 6:83-99

Naturskyddföreningen (2014) About us - Naturskyddföreningen http://www.naturskyddsforeningen.se/in-english/about-us. Accessed 21 August 2014

Estimated travel time to the nearest city of 50,000 or more people in year 2000 (2008) Global Environment Monitoring Unit - Joint Research Centre of the European Commission. http://bioval.jrc.ec.europa.eu/products/gam/. Accessed 07 November 2014

Newman DH, Wear DN (1993) Production economics of private forestry - A comparison of industrial and nonindustrial forest owners American Journal of Agricultural Economics 75:674-684 doi:10.2307/1243574

Ní Dhubháin Á, Cobanova R, Karppinen H, Mizaraite D, Ritter E, Slee B, Wall S (2007) The Values and Objectives of Private Forest Owners and Their Influence on Forestry

Behaviour: The Implications for Entrepreneurship Small-Scale Forestry 6:347-357 doi:10.1007/s11842-007-9030-2

Nichols P (1991) Social Survey Methods: A Fieldguide for Development Workers. Oxfam, Oxford, UK

Ninan KN, Inoue M (2013) Valuing forest ecosystem services: Case study of a forest reserve in Japan Ecosystem Services 5:78-87 doi:http://dx.doi.org/10.1016/j.ecoser.2013.02.006

Nkem J, Kalame FB, Idinoba M, Somorin OA, Ndoye O, Awono A (2010) Shaping forest safety nets with markets: Adaptation to climate change under changing roles of tropical forests in Congo Basin Environmental Science & Policy 13:498-508 doi:10.1016/j.envsci.2010.06.004

Nordlund A, Westin K (2011) Forest values and forest management attitudes among private forest owners in Sweden Forests 2:30-50 doi:10.3390/f2010030

North DC (1990) Institutions, institutional change and economic performance. Cambridge University press, Cambridge, UK

O'Hara KL (2006) Multiaged forest stands for protection forests: concepts and applications Forest Snow and Landscape Research 80:45-55

O'Neill BC et al. (2014) A new scenario framework for climate change research: the concept of shared socioeconomic pathways Clim Change 122:387-400 doi:10.1007/s10584-013-0905-2

Ojima DS, Galvin KA, Turner BL (1994) The global impact of land-use change Bioscience 44:300-304 doi:10.2307/1312379

Olabisi LS, Wang RQ, Ligmann-Zielinska A (2015) Why don't more farmers go organic? Using a stakeholder-informed exploratory agent-based model to represent the dynamics of farming practices in the Philippines Land 4:979-1002 doi:10.3390/land4040979

Ostrom E (2005) Understanding Institutional Diversity. Understanding Institutional Diversity. Princeton Univ Press, Princeton

Ostrom E (2009) A general framework for analyzing sustainability of social-ecological systems Science 325:419-422 doi:10.1126/science.1172133

Ostrom E, Burger J, Field CB, Norgaard RB, Policansky D (1999) Sustainability - Revisiting the commons: Local lessons, global challenges Science 284:278-282 doi:10.1126/science.284.5412.278

Ostry A, Ogborn M, Bassil KL, Takaro TK, Allen DM (2010) Climate Change and Health in British Columbia: Projected Impacts and a Proposed Agenda for Adaptation Research and Policy Int J Environ Res Public Health 7:1018-1035 doi:10.3390/ijerph7031018

Otte A, Simmering D, Wolters V (2007) Biodiversity at the landscape level: recent concepts and perspectives for multifunctional land use Landscape Ecol 22:639-642 doi:10.1007/s10980-007-9094-6

Pacheco JM, Vasconcelos VV, Santos FC (2014) Climate change governance, cooperation and self-organization Physics of Life Reviews 11:573-586 doi:http://dx.doi.org/10.1016/j.plrev.2014.02.003

Paquette S, Domon G (1999) Agricultural trajectories (1961-1991), resulting agricultural profiles and current sociodemographic profiles of rural communities in southern Quebec (Canada): A typological outline J Rural Stud 15:279-295 doi:10.1016/s0743-0167(98)90063-8

Pardos M, Calama R, Maroschek M, Rammer W, Lexer MJ (2015) A model-based analysis of climate change vulnerability of Pinus pinea stands under multiobjective management in the Northern Plateau of Spain Ann For Sci 72:1009-1021 doi:10.1007/s13595-015-0520-7

Parker DC, Manson SM, Janssen MA, Hoffmann MJ, Deadman P (2003) Multi-agent systems for the simulation of land-use and land-cover change: A review Ann Assoc Am Geogr 93:314-337 doi:Doi 10.1111/1467-8306.9302004

Parmesan C, Yohe G (2003) A globally coherent fingerprint of climate change impacts across natural systems Nature 421:37-42 doi:10.1038/nature01286

Pattberg P (2005) The institutionalization of private governance: How business and nonprofit organizations agree on transnational rules Governance-Int J Policy Adm 18:589-610 doi:10.1111/j.1468-0491.2005.00293.x

Philip LJ (1998) Combining quantitative and qualitative approaches to social research in human geography - an impossible mixture? Environ Plan A 30:261-276 doi:10.1068/a300261

Pierson P (2000) Increasing returns, path dependence, and the study of politics Am Polit Sci Rev 94:251-267 doi:10.2307/2586011

Plomion C et al. (2016) Forest tree genomics: 10 achievements from the past 10 years and future prospects Ann For Sci 73:77-103 doi:10.1007/s13595-015-0488-3

Preston BL et al. (2008) Mapping climate change vulnerability in the Sydney coastal group. Prepared for the Sydney Coastal Councils Group by the CSIRO Climate Adaptation Flagship.

Pukkala T, Kellomaki S (2012) Anticipatory vs adaptive optimization of stand management when tree growth and timber prices are stochastic Forestry 85:463-472 doi:10.1093/forestry/cps043

Purnomo H, Mendoza GA, Prabhu R, Yasmi Y (2005) Developing multi-stakeholder forest management scenarios: a multi-agent system simulation approach applied in Indonesia Forest Policy Econ 7:475-491 doi:DOI 10.1016/j.forpol.2003.08.004

R Development Core Team (2008) R: A language and environment for statistical computing. R Foundation for Statistical Computing. http://www.R-project.org. Accessed 02 November 2015

Railsback SF (2001) Concepts from complex adaptive systems as a framework for individual-based modelling Ecol Model 139:47-62 doi:10.1016/s0304-3800(01)00228-9

Ralha CG, Abreu CG, Coelho CGC, Zaghetto A, Macchiavello B, Machado RB (2013) A multi-agent model system for land-use change simulation Environ Modell Softw 42:30-46 doi:10.1016/j.envsoft.2012.12.003

Rammer W, Seidl R (2015) Coupling human and natural systems: Simulating adaptive management agents in dynamically changing forest landscapes Global Environmental Change-Human and Policy Dimensions 35:475-485 doi:10.1016/j.gloenvcha.2015.10.003

Rebaudo F, Dangles O (2015) Adaptive management in crop pest control in the face of climate variability: an agent-based modeling approach Ecol Soc 20 doi:10.5751/es-07511-200218

Reimann KD (2006) A view from the top: International politics, norms and the worldwide growth of NGOs Int Stud Q 50:45-67 doi:10.1111/j.1468-2478.2006.00392.x

Renwick A, Jansson T, Verburg PH, Revoredo-Giha C, Britz W, Gocht A, McCracken D (2013) Policy reform and agricultural land abandonment in the EU Land Use Policy 30:446-457 doi:10.1016/j.landusepol.2012.04.005

Richnau G et al. (2013) Multifaceted value profiles of forest owner categories in south Sweden: the River Helge a catchment as a case study Ambio 42:188-200 doi:10.1007/s13280-012-0374-2

Rico M, Gonzalez A (2015) Social participation into regional forest planning attending to multifunctional objectives Forest Policy Econ 59:27-34 doi:10.1016/j.forpol.2015.05.007

Roberge JM et al. (2016) Socio-ecological implications of modifying rotation lengths in forestry Ambio 45:S109-S123 doi:10.1007/s13280-015-0747-4

Robinson DT et al. (2007) Comparison of empirical methods for building agent-based models in land use science Journal of Land Use Science 2:31-55 doi:10.1080/17474230701201349

Ross-Davis A, Broussard S (2007) A typology of family forest owners in north central Indiana North J Appl For 24:282-289

Rounsevell MDA et al. (2014) Towards decision-based global land use models for improved understanding of the Earth system Earth Syst Dynam 5:117-137 doi:10.5194/esd-5-117-2014

Rounsevell MDA, Reay DS (2009) Land use and climate change in the UK Land Use Policy 26:S160-S169 doi:10.1016/j.landusepol.2009.09.007

Rounsevell MDA, Robinson DT, Murray-Rust D (2012) From actors to agents in socio-ecological systems models Philos T R Soc B 367:259-269 doi:DOI 10.1098/rstb.2011.0187

Rousseeuw PJ (1987) Silhouettes - A graphical aid to the interpretation and validation of cluster-analysis J Comput Appl Math 20:53-65 doi:10.1016/0377-0427(87)90125-7

Rudel TK, Coomes OT, Moran E, Achard F, Angelsen A, Xu JC, Lambin E (2005) Forest transitions: towards a global understanding of land use change Global Environmental Change-Human and Policy Dimensions 15:23-31 doi:10.1016/j.gloenvcha.2004.11.001

Rummukainen M, Gerger Swartling Å, Löwendahl E, Wallgren O, Hassler J, Smith B (2012) Mistra-SWECIA Annual Report 2011. Stockholm, Sweden

Rundqvist S, Hedenas H, Sandstrom A, Emanuelsson U, Eriksson H, Jonasson C, Callaghan TV (2011) Tree and shrub expansion over the past 34 years at the tree-line near Abisko, Sweden Ambio 40:683-692 doi:10.1007/s13280-011-0174-0

Rytter L, Karlsson A, Karlsson M, Stener L-G (2008) Skötsel av björk, al och asp. Skogsstyrelsen,

Satake A, Leslie HM, Iwasa Y, Levin SA (2007) Coupled ecological-social dynamics in a forested landscape: Spatial interactions and information flow Journal of Theoretical Biology 246:695-707 doi:10.1016/j.jtbi.2007.01.014

Schippers P et al. (2015) Landscape diversity enhances the resilience of populations, ecosystems and local economy in rural areas Landscape Ecol 30:193-202 doi:10.1007/s10980-014-0136-6

Schluter M et al. (2012) New horizons for managing the environment: a review of coupled social-ecological systems modelling Natural Resource Modeling 25:219-272 doi:10.1111/j.1939-7445.2011.00108.x

Schluter M, Pahl-Wostl C (2007) Mechanisms of resilience in common-pool resource management systems: an agent-based model of water use in a river basin Ecol Soc 12 doi:4

Schlyter P, Stjernquist I, Backstrand K (2009) Not seeing the forest for the trees? The environmental effectiveness of forest certification in Sweden Forest Policy Econ 11:375-382 doi:10.1016/j.forpol.2008.11.005

Schoene DHF, Bernier PY (2012) Adapting forestry and forests to climate change: A challenge to change the paradigm Forest Policy Econ 24:12-19 doi:10.1016/j.forpol.2011.04.007

Schroter D et al. (2005) Ecosystem service supply and vulnerability to global change in Europe Science 310:1333-1337 doi:10.1126/science.1115233

Scoones I (1998) Sustainable rural livelihoods: A framework for analysis vol Working Paper 72. IDS, Brighton

Sedjo RA (2010) Adaptation of forests to climate change: some estimates. Resources for the Future, Washington D.C.

Seidl R, Lexer MJ (2013) Forest management under climatic and social uncertainty: Trade-offs between reducing climate change impacts and fostering adaptive capacity J Environ Manage 114:461-469 doi:10.1016/j.jenvman.2012.09.028

Seppala R (2008) Global forest sector: trends, threats and opportunities. In: FreerSmith PH, Broadmeadow MSJ, Lynch JM (eds) Forestry and Climate Change. Cabi Publishing-C a B Int, Wallingford, pp 25-30

Sharma U, Patwardhan A (2008) An empirical approach to assessing generic adaptive capacity to tropical cyclone risk in coastal districts of India Mitigation and Adaptation Strategies for Global Change 13:819-831 doi:10.1007/s11027-008-9143-8

Siitonen J (2001) Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example Ecological Bulletins 49:11-41

Siry JP, Cubbage FW, Ahmed MR (2005) Sustainable forest management: global trends and opportunities Forest Policy Econ 7:551-561 doi:10.1016/j.forpol.2003.09.003

SLU (2010) SLU Forest Map. ftp://salix.slu.se/download/skogskarta. Accessed 03 March 2014

SLU (2015) Current data about the Swedish forests from the Swedish National Forest Inventory. http://www.slu.se/en/webbtjanster-miljoanalys/forest-statistics/skogsdata/. Accessed 08 April 2016

Smajgl A, Brown DG, Valbuena D, Huigen MGA (2011) Empirical characterisation of agent behaviours in socio-ecological systems Environ Modell Softw 26:837-844 doi:10.1016/j.envsoft.2011.02.011

SMHI (2016) Climate scenarios. http://www.smhi.se/en/climate/climate-scenarios#sc=rcp45. Accessed 03 October 2016

Smith B, Prentice IC, Sykes MT (2001) Representation of vegetation dynamics in the modelling of terrestrial ecosystems: comparing two contrasting approaches within European climate space Global Ecology and Biogeography 10:621-637 doi:10.1046/j.1466-822X.2001.t01-1-00256.x

Smith P et al. (2010) Competition for land Philos T R Soc B 365:2941-2957 doi:10.1098/rstb.2010.0127

Socolow RH (1999) Nitrogen management and the future of food: Lessons from the management of energy and carbon Proc Natl Acad Sci U S A 96:6001-6008 doi:10.1073/pnas.96.11.6001

Södra (2013) Timber and lumber products from Swedish oak. http://www.sodra.com/en/SodraTimberWorld/Our-products/Timber-and-lumber-products-from-birch-oak-and-alder/Timber-and-lumber-products-from-Swedish-oak/. Accessed 13 July 2015

Sohel MSI, Mukul SA, Burkhard B (2015) Landscape's capacities to supply ecosystem services in Bangladesh: A mapping assessment for Lawachara National Park Ecosystem Services 12:128-135 doi:10.1016/j.ecoser.2014.11.015

Sohngen B, Mendelsohn R, Sedjo R (1999) Forest management, conservation, and global timber markets American Journal of Agricultural Economics 81:1-13 doi:10.2307/1244446

Soja AJ et al. (2007) Climate-induced boreal forest change: Predictions versus current observations Glob Planet Change 56:274-296 doi:10.1016/j.glopacha.2006.07.028

Sorice MG, Kreuter UP, Wilcox BP, Fox WE (2014) Changing landowners, changing ecosystem? Land-ownership motivations as drivers of land management practices J Environ Manage 133:144-152 doi:10.1016/j.jenvman.2013.11.029

Spaargaren G, Mol APJ (2008) Greening global consumption: redefining politics and authority Global Environmental Change-Human and Policy Dimensions 18:350-359 doi:10.1016/j.gloenvcha.2008.04.010

Stern PC, Dietz T, Kalof L (1993) Value orientations, gender and environmental concern Environ Behav 25:322-348 doi:10.1177/0013916593255002

Sturtevant BR, Bissonette JA, Long JN, Roberts DW (1997) Coarse woody debris as a function of age, stand structure, and disturbance in boreal Newfoundland Ecological Applications 7:702-712

Swedish Board of Agriculture (2009) Facts about Swedish agriculture. Jönköping, Sweden

Swedish Environmental Protection Agency (2012) Sweden's environmental objectives - An introduction. Swedish Environmental Protection Agency, Stockholm

Swedish Environmental Protection Agency (2014a) Environmental objectives - Who does what? http://www.miljomal.se/sv/Environmental-Objectives-Portal/Undre-meny/Who-does-what/. Accessed 04/12 2014

Swedish Environmental Protection Agency (2014b) National Parks in Sweden - Naturvardsverket. http://www.swedishepa.se/Enjoying-nature/Protected-areas/National-Parks/. Accessed 20 August 2014

Swedish Environmental Protection Agency (2015) National inventory report Sweden 2015: Greenhouse gas emission inventories 1990-2013. Naturvårdsverket, Stockholm, Sweden

Swedish Environmental Protection Agency (2016) Milestone targets. http://www.swedishepa.se/Environmental-objectives-and-cooperation/Swedens-environmental-objectives/Milestone-targets/. Accessed 04 July 2016

Swedish Forest Agency (2014a) Organization. http://www.skogsstyrelsen.se/en/AUTHORITY/About-us/Organization/.

Swedish Forest Agency (2014b) Our Task - Skogsstyrelsen. http://www.skogsstyrelsen.se/en/AUTHORITY/About-us/Our-task/. Accessed 20 August 2014

Swedish Forest Agency (2014c) The Swedish Forestry Act. http://www.skogsstyrelsen.se/en/forestry/The-Forestry-Act/The-Forestry-Act/. Accessed 15 September 2014

Swedish Forest Agency (2015) Statistics - Subject Areas. http://www.skogsstyrelsen.se/en/AUTHORITY/Statistics/Subject-Areas/. Accessed 04 March 2015

Temme A, Verburg PH (2011) Mapping and modelling of changes in agricultural intensity in Europe Agric Ecosyst Environ 140:46-56 doi:10.1016/j.agee.2010.11.010

Temperli C, Bugmann H, Elkin C (2012) Adaptive management for competing forest goods and services under climate change Ecological Applications 22:2065-2077

Thompson JR, Foster DR, Scheller R, Kittredge D (2011) The influence of land use and climate change on forest biomass and composition in Massachusetts, USA Ecological Applications 21:2425-2444

Tilman D (1999) Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices Proc Natl Acad Sci U S A 96:5995-6000 doi:10.1073/pnas.96.11.5995

Tilman D et al. (2001) Forecasting agriculturally driven global environmental change Science 292:281-284 doi:10.1126/science.1057544

Tzilivakis J, Warner DJ, Green A, Lewis KA (2015) Adapting to climate change: assessing the vulnerability of ecosystem services in Europe in the context of rural development Mitigation and Adaptation Strategies for Global Change 20:547-572 doi:10.1007/s11027-013-9507-6

Ulmanen J, Swartling ÅG, Wallgren O (2012) Climate change adaptation in Swedish forestry - A historical overview, 1990-2010 vol Mistra-SWECIA Working Paper No 6. Stockholm Environment Institute,

E-rail traffic census in Europe 2005 (2015a) http://www.unece.org/trans/main/wp6/e-rails_census_2005.html. Accessed 15 October 2015

E-road traffic census in Europe 2005 (2015b) http://www.unece.org/trans/main/wp6/e-roads_census_2005.html. Accessed 15 October 2015

UNEP (2007) Global Environment Outlook 4: environment for development. United Nations Environment Programme, Valletta, Malta

Urquhart J, Courtney P (2011) Seeing the owner behind the trees: A typology of small-scale private woodland owners in England Forest Policy Econ 13:535-544 doi:http://dx.doi.org/10.1016/j.forpol.2011.05.010

Valbuena D, Verburg PH, Bregt AK (2008) A method to define a typology for agent-based analysis in regional land-use research Agric Ecosyst Environ 128:27-36 doi:10.1016/j.agee.2008.04.015

Valbuena D, Verburg PH, Bregt AK, Ligtenberg A (2010) An agent-based approach to model land-use change at a regional scale Landscape Ecol 25:185-199 doi:DOI 10.1007/s10980-009-9380-6

Valladares F (2008) A mechanistic view of the capacity of forests to cope with climate change vol 17. Managing Forest Ecosystems. doi:10.1007/978-1-4020-8343-3_2

Van Asselen S, Verburg PH (2013) Land cover change or land-use intensification: simulating land system change with a global-scale land change model Glob Change Biol 19:3648-3667 doi:10.1111/gcb.12331

Van Berkel DB, Verburg PH (2012) Combining exploratory scenarios and participatory backcasting: using an agent-based model in participatory policy design for a multi-functional landscape Landscape Ecol 27:641-658 doi:10.1007/s10980-012-9730-7

Van Bogaert R, Jonasson C, De Dapper M, Callaghan TV (2010) Range expansion of thermophilic aspen (Populus tremula L.) in the Swedish Subarctic Arctic Antarctic and Alpine Research 42:362-375 doi:10.1657/1938-4246-42.3.362

van Gameren V, Zaccai E (2015) Private forest owners facing climate change in Wallonia: Adaptive capacity and practices Environmental Science & Policy 52:51-60 doi:10.1016/j.envsci.2015.05.004

van Vuuren DP et al. (2011) The representative concentration pathways: an overview Clim Change 109:5-31 doi:10.1007/s10584-011-0148-z

van Vuuren DP et al. (2014) A new scenario framework for Climate Change Research: scenario matrix architecture Clim Change 122:373-386 doi:10.1007/s10584-013-0906-1

Världsnaturfonden (2014) Världsnaturfonden WWF Sverige. http://www.wwf.se/. Accessed 21 August 2014

Venier LA, Dalley K, Goulet P, Mills S, Pitta D, Cowcill K (2015) Benefits of aggregate green tree retention to boreal forest birds Forest Ecology and Management 343:80-87 doi:10.1016/j.foreco.2015.01.024

Vereijken PH (2002) Transition to multifunctional land use and agriculture Neth J Agric Sci 50:171-179

Verkerk PJ, Mavsar R, Giergiczny M, Lindner M, Edwards D, Schelhaas MJ (2014) Assessing impacts of intensified biomass production and biodiversity protection on ecosystem services provided by European forests Ecosystem Services 9:155-165 doi:10.1016/j.ecoser.2014.06.004

Villamor GB, Le QB, Djanibekov U, van Noordwijk M, Vlek PLG (2014) Biodiversity in rubber agroforests, carbon emissions, and rural livelihoods: An agent-based model of land-use dynamics in lowland Sumatra Environ Modell Softw 61:151-165 doi:10.1016/j.envsoft.2014.07.013

Vincent K (2007) Uncertainty in adaptive capacity and the importance of scale Global Environmental Change-Human and Policy Dimensions 17:12-24 doi:10.1016/j.gloenvcha.2006.11.009

Vulturius G, André K, Gerger Swartling Å, Brown C, Rounsevell MDA, Jönsson AM, Blanco V (in review) Explaining engagement with climate change adaptation among forest owners in Sweden. Submitted to Regional Environmental Change

Wang J, Brown DG, Riolo RL, Page SE, Agrawal A (2013) Exploratory analyses of local institutions for climate change adaptation in the Mongolian grasslands: An agent-based modeling approach Global Environmental Change-Human and Policy Dimensions 23:1266-1276 doi:10.1016/j.gloenvcha.2013.07.017

Wang YF, Vasilakos AV, Ma JH, Xiong NX (2015) On studying the impact of uncertainty on behavior diffusion in social networks Ieee Transactions on Systems Man Cybernetics-Systems 45:185-197 doi:10.1109/tsmc.2014.2359857

Werland S (2009) Global forest governance - Bringing forestry science (back) in Forest Policy Econ 11:446-451 doi:10.1016/j.forpol.2008.07.002

Wiersum KF, Elands BM, Hoogstra M (2005) Small-scale forest ownership across Europe: Characteristics and future potential Small-scale Forest Economics, Management and Policy 4:1-19 doi:10.1007/s11842-005-0001-1

Wiggering H et al. (2006) Indicators for multifunctional land use - Linking socio-economic requirements with landscape potentials Ecol Indic 6:238-249 doi:10.1016/j.ecolind.2005.08.014

Work TT, Jacobs JM, Spence JR, Volney WJ (2010) High levels of green-tree retention are required to preserve ground beetle biodiversity in boreal mixedwood forests Ecological Applications 20:741-751 doi:10.1890/08-1463.1

Worldwatch Institute (2012) State of the world 2012: Moving toward sustainable prosperity. Washington, DC

Wossen T, Berger T (2015) Climate variability, food security and poverty: Agent-based assessment of policy options for farm households in Northern Ghana Environmental Science & Policy 47:95-107 doi:10.1016/j.envsci.2014.11.009

Wu WB, Yu QY, Peter VH, You LZ, Yang P, Tang HJ (2014) How could agricultural land systems contribute to raise food production under global change? Journal of Integrative Agriculture 13:1432-1442 doi:10.1016/s2095-3119(14)60819-4

Yang XG et al. (2015) Potential benefits of climate change for crop productivity in China Agricultural and Forest Meteorology 208:76-84 doi:10.1016/j.agrformet.2015.04.024

Yokomizo H, Coutts SR, Possingham HP (2014) Decision science for effective management of populations subject to stochasticity and imperfect knowledge Popul Ecol 56:41-53 doi:10.1007/s10144-013-0421-2

Zanchi G, Pena N, Bird N (2012) Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel Global Change Biology Bioenergy 4:761-772 doi:10.1111/j.1757-1707.2011.01149.x

Zhang PC, Shao GF, Zhao G, Le Master DC, Parker GR, Dunning JB, Li QL (2000) Ecology - China's forest policy for the 21st century Science 288:2135-2136 doi:10.1126/science.288.5474.2135

Appendices

Appendix A

Table A.1 Studies that contributed to the creation of the forest manager typology

Study	Type of Study	Relevant Information	Country/ Country cluster	Geographical Scale
Amacher et al. (2003)	Review	NIPF* owner decision-making	-	-
Andersson (2012)	Original article	NIPF socio-demographic/ economic attributes	Sweden	Subnational
Arano and Munn (2006)	Original article	Forest owner management intensity	United States	State
Beach et al. (2005)	Review	NIPF owner management	-	-
Boon et al. (2004)	Original article	Private forest owner typology	Denmark	National
Canadas and Novais (2014)	Original article	NIPF owner socio-economic attributes and management	Portugal	National
Conway et al. (2003)	Original article	NIPF owner decision-making	United States	Subnational
Creighton et al. (2002)	Original article	NIPF ownership and ecosystem management	United States	State
Duncker et al. (2012)	Original article	Classification of forest management approaches	-	-
Eggers et al. (2014)	Original article	NIPF owner attributes and their management strategies	Sweden	National
Emtage et al. (2007)	Original article	Landowner typologies in supporting policy	-	-
Eriksson (2012)	Original article	Private forest owner values and beliefs	Sweden	National
Finley and Kittredge (2006)	Original article	Private forest owner management and typology	United States	State
Fujimori (2001)	Book	Forest management strategies	-	-
Hogl et al. (2005)	Original article	Private forest owner characteristics and typology	Austria	National
Ingemarson et al. (2006)	Original article	NIPF owner typology	Sweden	National
Joshi and Arano (2009)	Original article	Determinants of private forest management decisions	United States	State
Karppinen (1998)	Original article	NIPF owner values, objectives and typology	Finland	Subnational

Kline et al. (2000)	Original article	Forest owner objectives and socio-economic characteristics	United States	Multistate
Kvarda (2004)	Original article	NIPF owner objectives	Austria	Subnational
Liao and Zhang (2008)	Original article	Econometric comparison of industrial and NIPF owners	United States	Multistate
Lidestav (2010)	Original article	Forest property inheritance	Sweden	National
Lönnstedt (2012)	Original article	NIPF owner motives for ownership	Sweden	Subnational
Majumdar et al. (2008)	Original article	NIPF owner motivations and typology	United States	Multistate
Newman and Wear (1993)	Original article	Production economics of industrial and NIPF owners	United States	Multistate
Ní Dhubháin et al. (2007)	Review	Private forest owner values and objectives	-	-
Nordlund and Westin (2011)	Original article	Private forest owner values and management attitudes	Sweden	National
Ross-Davis and Broussard (2007)	Original article	NIPF owner typology	United States	Subnational
Stern et al. (1993)	Original article	Value orientations, gender and environmental concern	-	-
Urquhart and Courtney (2011)	Original article	NIPF owner typology	United Kingdom	Subnational
Wiersum et al. (2005)	Original article	Small-scale forest owner characteristics and typology	European Union (8 countries)	Supranational

* Non-Industrial Private Forest

Appendix B

ODD Protocol for CRAFTY-Sweden

The model description follows the ODD (Overview, Design concepts, Details) protocol for describing individual- and agent-based models (Grimm et al. 2006; Grimm et al. 2010). Some repetition may exist between this document and thesis chapters, as the ODD Protocol is meant to be stand-alone. This ODD Protocol was largely developed by Sascha Holzhauer (University of Edinburgh), with input from me.

1 Purpose

The 'Competition for Resources between Agent Functional Types in Sweden' (CRAFTY-Sweden) model is designed to model land-use changes in Sweden, with a focus on forestry. CRAFTY-Sweden applies the model framework, CRAFTY-CoBRA which is an extension of CRAFTY (Murray-Rust et al., 2014) that allows for dynamic production functions. CRAFTY can be used to investigate the effects of human behaviour on land use transitions under a range of socio-economic and environmental scenarios. CRAFTY is designed to be flexible, capable of handling a large variety of data and to be applicable across a wide range of empirical or theoretical settings.

CRAFTY-Sweden is founded on efficient and tractable descriptions of individual behaviour and decision-making that takes account of the effects of climatic and environmental change, and may be adapted to a range of applications and scenarios. It applies exogenous demand levels, which agents attempt to meet according to behavioural rules and ecosystem service supply. The model considers the adoption of different land uses, variations in the intensity of land uses, diversification into multifunctional land uses, changes in productivity over time, land abandonment, and competition for available land.

Agents use capitals that are available for the land parcels that they own and supply ecosystem services based on their respective production function. Capitals may vary across the modelled landscape and over time. Agents of the same functional role may have the same or heterogeneous weights for the production function and this together with the available capitals determines the supply level for each service (i.e., timber for different forest types, food, biodiversity, carbon sequestration, and recreation) by the agents. The current demand for a particular service and an agent's productivity determine its competitiveness, which in turn, affects the introduction of new agents in the model: the distribution of agents over a landscape and the introduction of new agents into the system during simulation are determined by an 'allocation procedure', which is discussed in section *7 Submodels*, Allocation. Institutions may change capital levels and issue land use restrictions (see section *7 Submodels*, Institutions).

2 Entities, state variables, and scales

Spatial units. CRAFTY-Sweden is based on a grid of *cells*, representing any absolute spatial scale. Each cell has defined levels of a range of *capitals*, which describe the availability of particular social, environmental or economic resources. A non-spatial population is assumed to exist and to generate demands for *services*. Each cell may be managed by a single land use *agent.*

Agents. Forest owners and farmers are explicitly represented as agents in CRAFTY-Sweden. Both share a common architecture where agents are made up of a functional role (FR) characterising function and role in the system, and a collection of properties.

A **land use agent** is able to leverage the *capitals* available on a land parcel (represented as a *cell*) to provide a range of *services*. Each agent has a production function as part of its FR, which maps capital levels onto service provision (see section *7 Submodels*, Production). An agent's *competitiveness*, according to a given level of service provision, can be calculated from societal demands, overall supply levels and marginal utility functions. See Table B.1 for a complete list of agent variables.

Table B.1 Variables and States of agents and their FR. Default states are given in parenthesis if applicable.

Variable	***Description***	***States***
Competitiveness	Denotes the agent's current competitiveness value	]-∞,∞[

Giving-in threshold	During competition, if a competing agent's competitiveness is greater than the incumbent agent's by a value larger than the giving-in threshold then the incumbent agent relinquishes that cell to the competitor.	]-∞,∞[
Giving-up threshold	If an agent's competitiveness falls below its giving-up threshold (defines the minimum return an agent is willing to accept from a cell) it needs to abandon the particular cell (considering giving-up probability).	]-∞,∞[
Giving-up probability	Probability for giving up in case the agent's competitiveness falls below the giving-up threshold	[0,1] (1.0)
Functional component		
Role	Refers to the Functional Role (FR)	Reference
Optimal production	Amount of produced service in case of optimal conditions (all relevant capitals = 1.0)	[0, ∞[or formula (based on JEP[3])
Capital sensitivities	Sensitivities of production towards capital values	[0,1] (1.0)
Production model	Component responsible for the calculation of service provision	SimpleProductionModel or DynamicMaxProductionModel

An agent searching for land can either take over unmanaged (abandoned) cells, or cells on which it can outcompete an existing agent. Between them, giving-up and giving-in parameters provide a stylised interpretation of factors that make human behaviour deviate from narrowly defined optimality, such as personal connection to a landscape or way of life, or resistance to change.

The functional roles (FR) that are assigned to agents make up a typology that defines general characteristics of land manager practices. Searching agents can be prototypes of specific FRs that allow the comparison of productivity, utility and other characteristics of "typical" agents of that FR. Finally, individual agents of a given type need not be identical – all of the agent's characteristics, including production functions, and giving up/giving in thresholds are drawn from distributions to provide within-type heterogeneity.

I developed a typology of forest and agricultural agents, focusing especially on the former. To define forest owner types I used as a basis the forest owner typology developed in Chapter

[3] Java Expression Parser (http://www.cse.msu.edu/SENS/Software/jep-2.23/doc/website/index.html)

2. Because this typology was based on studies performed at different scales and contexts to those found in Sweden, I performed a validation exercise of the typology using empirical information from 872 Swedish forest owners (Vulturius et al. in review). A cluster analysis showed that the five overarching management roles identified by the theoretical typology (productionist, multi-objective, recreationalist, conservationist and passive) were also clearly discernible in the empirical data. Supplementary materials on this validation can be found in Appendix B.1.

Within each overarching management role, different options for forest management are possible, including the use of different types of forest (defined by species composition). Forest types were assigned to each management role on the basis of existing forest stand compositions (Swedish Forest Agency 2015) and potential adaptation measures to climate change that consider species composition, number of thinnings and rotation lengths (Felton et al. 2016; Jonsson et al. 2015) on the basis of owner objectives (Chapter 2; (Duncker et al. 2012a). Assigned forest types were pine (*Pinus sylvestris*), spruce (*Picea abies*), boreal broadleaf (*Betula pendula, B. pubescens, Alnus incana, A. glutinosa, Populus tremula*), nemoral broadleaf (*Fagus sylvatica, Quercus robur, Fraxinus excelsior, Ulmus glabra, Tilia cordata, Carpinus betula*), and combinations of these, resulting in 17 forest owner types.

Given the current levels of agricultural production (Swedish Board of Agriculture 2009) and management intensities prevailing in Sweden (Institute of Environmental Studies 2015), farmers were separated by the main services provided (i.e. cereal or meat) in combination with their main objectives (i.e. commercial or non-commercial).

***Environment*.** The environment represents the terrain of Sweden with its varying geophysical features. This heterogeneity in the modelled landscape is represented by the amount of capitals (such as economic, nature, infrastructure) that exists on a cell. For instance, a cell in a forest region will have much higher natural capital than infrastructural capital.

Scales. A time step in CRAFTY-Sweden represents a year in practice since this is the time span for which land managers make decisions, which is true at least for farmers who grow perennial crops. Space is represented by a grid of $1km^2$ cells.

3 Process overview and scheduling

At each modelled time step, the level of service production achieved by an agent is given a benefit value via a benefit function that relates production levels to unmet demand. Agents compete for land based on these benefit values, and this competition is affected by individual or typological behaviour. Table B.2 gives an overview of the CRAFTY-Sweden simulation schedule.

Table B.2 Simulation schedule

```
For each agent ∈ agents
  increase age
For each agent ∈ agents
  update competitiveness based on demand_residual
  If competitiveness < threshold_giving-up
      leave cell
For each region ∈ regions
  allocate land: for each unmanaged cell
      allocate new agent of most competitive functional role
  compete for land: for each fr ∈ functional_roles
      calculate fr's competitiveness on perfect cell
  For n search iterations
      select fr randomly according to competitiveness
      For m cells
          calculate fr's competitiveness
          If fr's competitiveness > owner's threshold_giving-in
              owner relinquishes cell
              agent with fr takes over cell
For each agent ∈ agents
  update supply of services produced
For each region ∈ regions
  update supply and demand_residual
For each agent ∈ agents
  update competitiveness based on demand_residual
generate output
```

Each time step starts by updating the decision-making context for land use agents – the levels of demand, capitals and any active policies. This has two stages:

- Updates are made to the levels of demand across each region, and levels of capitals within each cell. These are typically loaded from external files, either as direct values or as functions to be sampled from on a yearly basis. Mechanisms are also available to dynamically modify capitals, for example in order to model land degradation through intensive agriculture, allowing for feedback loops in this SES.

Next, the land use agents respond and adapt to this altered context:

- First, each agent updates its level of supply, based on current capital levels. The total supply of each service is then calculated.
- Next, each agent's competitiveness is calculated, including the effect of any institutional policies.
- Any agents who give up are removed from the model.
- The active allocation procedure now runs, allowing new agents to take over unmanaged land and allowing other land transitions to take place, subject to restrictions for certain transitions.

Once all of the land use agents have been updated, final accounting is carried out, such as calculating total supply and demand, creating output files, displaying model state and creating model run animations.

4 Design concepts

Basic principles. The design criteria used for the specification of the model framework were:

1) The model must be able to run at large scales. This requirement holds for runtime costs, complexity, and the availability of data to parameterise and calibrate the model.
2) The model should take into account the full range of societal demands, including those that are not defined explicitly in monetary terms such as biodiversity.
3) The model must be able to represent multifunctional land use, and be responsive to the trade-offs between the provision of various services.
4) The model should be able to represent the diversity of human behaviour and land management.
5) The model must be able to deal with the long-term allocation of forest types.

Agent Functional Types are derived from the concept of Plant Functional Types in Dynamic Vegetation Models (e.g. Lavorel et al. 2007) and used to group land-use agents by their decision making and productive behaviour; here adopted as Functional Roles. Land-use modellers are familiar with the use of typologies, especially in constructing agent-based models as representations of real-world actors (Robinson et al. 2007; Valbuena et al. 2008). Typologies allow generalisations of the attributes (traits) of individual actors in a system that simplifies model development and application, and provides a more transparent representation of agent decisional processes and behaviour.

Table B.3 provides an overview of the assumptions that guided the model framework development.

Table B.3 Design assumptions made in CRAFTY-Sweden

Model assumption	**Details**	**Justification**
A wide range of land-use relevant behaviour can be represented by 'giving-in' and 'giving-up' thresholds.	Range of personal characteristics and behaviours known to affect land use decisions can be often abstracted in two values giving (relative) willingness of land managers to change land use or abandon land. Believed to be a necessary simplification for large-scale land use models that adequately mimics observed behaviour but can be 'overwritten' by more specific decisions (see sections 2 Entities, state variables, and scales, Agents and 7 Submodels, Decision Making).	Known that numerous factors affect personal decision-making (e.g. Siebert *et al.* 2006; Gorton *et al.* 2008; Valbuena *et al.* 2010; Meyfroidt 2012) - too many to model or parameterise. Several studies have suggested that, for modelling purposes, a wide range of behaviours are reducible to a small number of dimensions similar to those used here (e.g. Berger 2001; Polhill et al. 2001; Siebert et al. 2006; Gorton et al. 2008; Murray-Rust et al. 2011).
Each cell is managed by a single agent.	Multiple ownership of cells is not supported.	The scale of application is not defined and so can be set to the appropriate scale of land holdings in any particular case (the minimum size of holding that is of interest to the modeller). Agents may be permitted to manage multiple cells.
Land managers can be Functional Roles (FR).	The management practices (FR) and behaviour (BT) of land managers allows them to be classified into a typology analogous to the Plant Functional Types used in Dynamic Global Vegetation Models, increasing modelling efficiency.	The use of types increases computational efficiency by providing a description of land management and human behaviour at a level of abstraction that decreases the need for empirical parameterisation but retains the characteristics most important to large-scale land use change (Arneth et al. 2014). Splitting in FR and BT allows changes in one component while the other persists.
Potential productivity of land can be represented by a range of capitals.	Capitals representing natural productivity (for any good or service such as a specific food or timber) and any anthropogenic effects on productivity (such as availability of finance or infrastructure) can be used as a basis for the description of ecosystem services.	Well-established method of characterising land – see Boumans et al. (2002) and Scoones (1998).
Production of services by land managers can be described by a	The ability of land managers to produce services is dependent on the underlying productivity of the land, expressed via	Douglas (1976), Fulginiti and Perrin (1998) and Martin and Mitra (2001)

function dependent upon access to capitals and productive abilities.	capitals (above) and their individual or typological productive ability, which may depend upon a number of personal characteristics and behavioural factors. (A Cobb-Douglas function is used, adapted to incorporate a time component for forest services.)	
The competitiveness of land managers depends upon demand for specific services.	Demands exist for ecosystem goods and services, and land managers compete to satisfy these demands. Land managers are more competitive when they can produce greater (total) quantities of services for which there is unmet demand.	Demands for services are known to be expressed via the economic value of service production, and, in the absence of behavioural factors, land use is driven primarily by economics. Partly, decisions are made on grounds of non-monetary (or indirectly monetary) demands – e.g. for green space, fresh water etc. – and CRAFTY-Sweden is designed to be capable of handling these, where they can be parameterised. No assumption about the relationship between unmet (residual) demand and utility values (competitiveness) is made.
Three mechanisms of land use change.	Land use (or ownership) changes when agents abandon land, take over unmanaged land, or take over managed land from the current owner.	The same limited number of options are possible in the real world.
Lock forest management approach until the forest has reached maturity.	Forest owners will not abandon or change the management approach on their land until the forest has reached maturity, in order to recover the initial investment.	Based on expert knowledge from Swedish forestry researchers
A forest owner will not fell a forest before a minimum felling age.	A forest owner will not fell a forest before a minimum felling age dependent on site quality (i.e. productivities), which is determined by law in Sweden for pine and spruce, and for which recommendations exist for other species.	Forests are felled in Sweden after reaching an age that depends on site quality (Lagergren et al. 2012). The stand age at felling is regulated in law for pine and spruce to guarantee that the production potential is utilised (Kunskap Direkt 2015), and for beech, birch and oak recommended rotation periods exist (Löf et al. 2009; Rytter et al. 2008).
Passive owners do not plant their forest, and their production is dependent on the management of the agent they take over the forest from.	They therefore only take over the forest and associated optimal production function of other owner types managing forests with the same tree species as them, but do decide about the forest age at felling.	Passive owners' generalised lack of primary objectives for forestry (Chapter 2).

Emergence. Emergent effects that could be observed as outcomes of experiments using CRAFTY-Sweden are spatially explicit changes to land ownership and management, the intensification of land uses, including mono- or multi-functional land uses, changes in productivities and yields of different land uses, and effects on capital levels.

Adaptation**.** Land use agents may change their functional role in response to unsatisfactory competitiveness, e.g., due to changes in demands for the service they produced so far.

Objectives. The land manager's objective is to be sufficiently competitive in the supply of societal benefits.

Learning. There is no learning in CRAFTY-Sweden.

Fitness. Agents' survival in the system depends upon their *competitiveness*, which is determined by an agent's ability to meet the demand of services in a modelled society (see Section 7, Population, Services, Demand and Utility).

Prediction**.** Agents in CRAFTY-Sweden do not explicitly predict.

Sensing**.** Land use agents in CRAFTY-Sweden are aware of the current demand (regional or global depending upon the setup) that is to be met. They consider the production they are able to achieve on a given cell with a particular capital level, e.g., when they want to change their functional role. Agents are aware of the competitiveness of other agents and may relinquish their cells to agents that are more competitive.

Interaction**.** Direct interactions occur between new ('potential') and existing agents that compete for cell ownership. Land owner agents compete for land on the basis of their benefit values, which depend upon their ability to produce services and societal demand levels for those services. As land ownership and management change, demand and supply levels also change, so that actions taken by each agent affect the decisions of others.

Stochasticity. There are a number of stochastic processes in CRAFTY-Sweden:

- Initialisation of agent properties (e.g., giving-in/up thresholds, optimal production, capital sensitivities) from probabilistic distributions and/or the addition of noise
- Search for cells during competition
- Selection of functional roles to compete with incumbent land managers
- Probabilistic giving-up
- Forest stand felling age

Random numbers are sourced from different random streams which allows their separate control via defined initial random seeds (see CRAFTY online documentation[4]). This also ensures the reproducibility of simulations results.

Collectives. Two types of agent collectives exist during a course of a simulation run. First is the list of agents that possess land parcels (cells) in the simulated landscape (grid). Second is the list of potential agents that enter the system to takeover cells from existing agents (if possible) or occupy a vacant or abandoned cell on the grid.

***Observation*.** CRAFTY provides a range of observations and displays to help understand the model behaviour during runtime. Each of the submodels has a display, which is either numeric or graphical, showing curves for variables of note. A range of spatially explicit outputs is also available; these include agent type, capital levels, competitiveness scores, supply of services, and so on. Any of these displays can be used to create videos of the model's behaviour over time.

Furthermore, CRAFTY enables the output of a number of simulation data as CSV or raster files.

5 Initialization

To allow the framework's configuration by non-programmers it is accomplished by a set of interlinked XML and CSV files. XML files define basic simulation parameters and provide properties for the initialisation of model components coded as Java objects, while CSV files provide data when there are many values required. The approach is highly flexible and extendable.

CRAFTY-Sweden is initialised by reading the file Scenario.xml and following the links therein to the configuration of outputs and the world configuration, which in turn contains links to model components such as functional roles, the competition model, or the allocation model. A file Cells.csv includes the coordinates and capital levels of the cells in a region, the initial allocation of agents on these cells, and agent's properties such as functional role, which are

[4] https://www.wiki.ed.ac.uk/display/CRAFTY/Randomisation

applied when these agents are initialised. Fig. B.1 gives an overview of a possible setting of XML and CSV files.

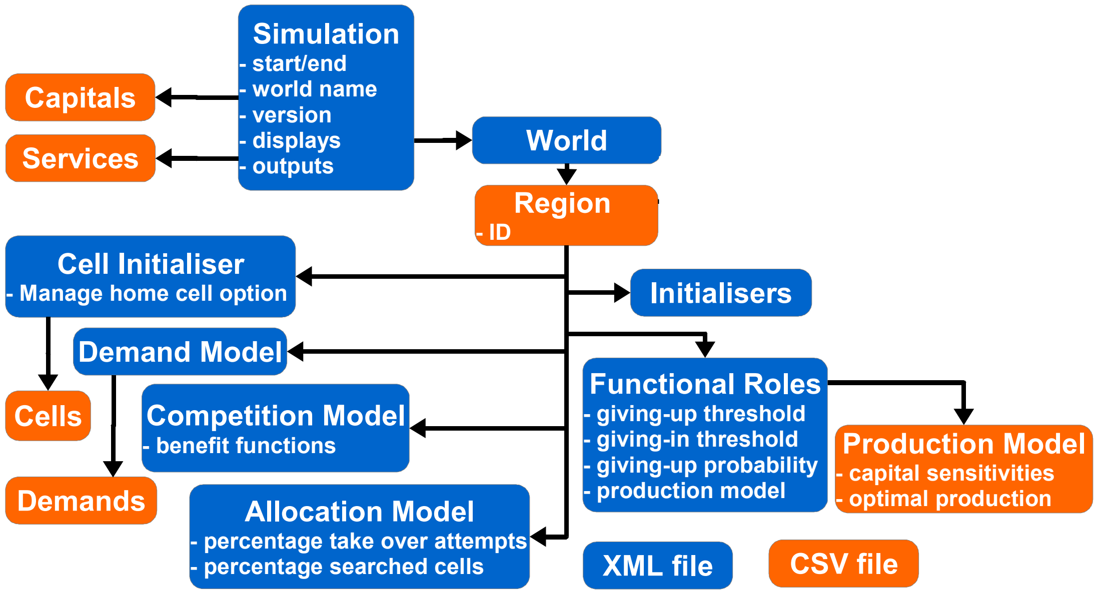


Fig. B.1 Overview of model configuration, showing relationships between files and what each file provides

6 Input data

Land cover data. To create a baseline land ownership map for 2010 I first devised a land-use map at 1km^2 resolution that included pine, spruce, pine-spruce, pine-boreal broadleaf, spruce-boreal broadleaf, boreal broadleaf, and nemoral broadleaf productive forests, agriculture, protected areas, non-productive forests, semi-natural vegetation, wetlands, open spaces, 'other unmanaged' land, artificial, and water bodies. SLU Forest Map data (SLU 2010) on the proportion of different tree species per cell were used to identify forest cover and classify it according to the forest types assigned above, according to the proportion of forest within the cell, and the proportions of different species within that forest. CORINE land cover (EEA 2014) was used to identify all other land use/land cover classes. Nationally Designated Areas (EEA 2015) were then superimposed to define protected areas. Non-productive forests are also protected and unavailable for production (Swedish Forest Agency 2014c). Thus, I identified them by:

3. Assigning to forested cells the value of the highest productivity found among all forest types within that cell; and
4. Given the proportion of non-productive forest per county (Swedish Forest Agency 2015), selecting for each county the equivalent number of cells with the lowest productivity values.

Mean forest age values from the SLU Forest Map were used to assign forest ages.

Forest owner types were allocated to productive forest types using data on a) the area of productive forest land by county and ownership classes for 2010 (Swedish Forest Agency 2015); and b) the proportion of owners in each county belonging to each group from the cluster analysis. Agricultural land and (some) semi-natural vegetation were assigned to commercial cereal, non-commercial cereal, commercial livestock, and non-commercial livestock farmer agents according to the land-use intensity in 2010 (Institute of Environmental Studies 2015). The remaining semi-natural vegetation, wetlands, and 'other unmanaged' land were left unallocated. Protected areas, non-productive forests, open spaces with little or no vegetation, artificial surfaces, and water bodies were made unavailable for allocation during simulations. Further detail on the creation of land-use and land owner type maps can be found in Appendix B.3.

Capital levels. The capitals that agents can use in service production are productivities for pine, spruce, boreal broadleaf, and nemoral broadleaf forests, grassland productivity (natural capital), and transportation infrastructure (infrastructure capital). Table B.4 gives capital descriptions, their data sources, and the ecosystem services they contribute to producing.

Land productivity levels can be affected by climate change. The ecosystem model LPJ-GUESS (Smith et al. 2001) was used to simulate forest dynamics during 2010-2100 using climate projections of the Global Circulation Model-Regional Circulation Model ensembles (hereupon 'climate models') EC-Earth-RCA4, IPSL-RCA4 and NorESM-RCA4 for RCPs 4.5 and 8.5 from the EURO-CORDEX project (Jacob et al. 2014; Jones et al. 2011). Annual climate-induced change was calculated for all productivities using LPJ-GUESS spatial projections of yearly timber volume growth for pine, spruce, boreal broadleaf and nemoral broadleaf forests, and yearly net primary productivity (NPP) change for grass until 2100 at 50x50 km resolution. After checking for non-linearities in the volume growth and NPP change projections, linear models were considered to be sufficient to represent them. Thus, a regression coefficient was calculated for every cell by performing linear regression on projected growth values. These values were then downscaled to 1 km^2. See Appendix B.6 for more details about the calculation of climate impacts on productivities.

Table B.4 Identities and data sources for modelled capitals, and the ecosystem services they contribute to producing

CAPITAL	DEFINITION	INPUT DATA (units; resolution)	ECOSYSTEM SERVICES	DATA SOURCE
Pine, Spruce, Boreal Broadleaf, and Nemoral Broadleaf Forest Productivities	Baseline productive potential for each forest type	Forest production potential per forest type (m^3sk ha^{-1} yr^{-1}; 1km^2)	- Pine, spruce, boreal br. and nemoral br. timber - Carbon - Biodiversity	(Hägglund and Lundmark 1987) (Johansson et al. 2013) SLU
Grassland Productivity	Baseline productive potential for grassland and cropland	LPJ-GUESS simulated C3-grass NPP projection for 2010 driven by climate (radiation, temperature, precipitation) (kg C m^{-2} yr^{-1}; 50x50 km)	- Meat - Cereal	Simulations done for this article (see section 2.2)
Transportation Infrastructure	Proximity to transportation networks and central markets	1. Road and rail networks 2. Waterway networks 3. Travel time to nearest town with over 50000 inhabitants (1km^2)	- Pine, spruce, boreal br. and nemoral br. timber - Meat - Cereal - Recreation	1. UNECE 2. EEA 3. GEMU, JRC

SLU: SLU Forest Map, produced by Swedish University of Agricultural Sciences, accessed via ftp://salix.slu.se/download/skogskarta
UNECE: United Nations Economic Commission for Europe, accessed via http://www.unece.org/trans/areas-of-work/transport-statistics/statistics-and-data-online.html
EEA: European Environment Agency, accessed via http://www.eea.europa.eu/data-and-maps
GEMU, JRC: Global Environment Monitoring Unit, managed by the Joint Research Centre, accessed via http://www.edenextdata.com/?q=content/jrc-accessibility-map-estimated-travel-time-nearest-city-population-50000

Demand levels. Demand levels (for ecosystem services) are exogenous to the model and are defined prior to model initialisation based on land-use and an interpretation for Europe of the storylines of the Shared Socio-economic pathways (SSPs) (Carter et al. 2015) from Engstrom et al (2016) and Kok et al. (2015) respectively. Quantifications of demands were done for SSP1, SSP3, SSP4, and SSP5. Baseline demands for timber, cereal and meat were assumed to be equal to observed production in 2010 (FAO 2015; Swedish Forest Agency 2015), while those for carbon sequestration, biodiversity and recreation were assumed equal to simulated baseline supply due to lack of empirical data.

Future projections were calculated using the IIASA SSP data (IIASA 2015) on decadal rates of change of global forest land cover (for timber and carbon sequestration), and crop and livestock demands. Demands for biodiversity were projected following the SSP storylines and with guidance from modelled global future changes in species abundance from UNEP (2007). Rates of change in recreation demands were assumed to be the same as those for biodiversity.

7 Submodels

Allocation Model. Land ownership within CRAFTY-Sweden changes according to three different mechanisms, which simulate both individual and collective aspects of land use dynamics. Firstly, agents may abandon their land owing to the competitiveness score that falls below an agent's giving-up threshold.

Secondly, when land is unmanaged, due to abandonment or lack of managers, it can be taken over by a newly created agent. By default, the set of functional roles is evaluated to determine agent competitiveness score on each unmanaged cell ($c_{a,i}$). The functional roles are sampled such that the probability of a role *a* attempting to take over a cell scales with its competitiveness on a cell with 'perfect' capital levels; $P(a) \propto {c_{a,i}}^{\gamma}$, where γ=0 gives a random selection and $\gamma \to \infty$ tends towards optimal selection.

For more general land use transitions, an allocation procedure runs between active and potential or ambulant agents to determine ownership changes. This can include direct competition, where incoming agents attempt to take over existing cells; such an attempt succeeds where the new agent has a competitiveness on the cell greater than or equal to the existing agent's competitiveness plus its giving-in threshold: $c_{new} \geq c_{curr} + giving\_in_{curr}$.

Production function. The production of agricultural services is modelled on a yearly basis. Forestry services are however dependent on forest age. Additionally, climatic change can affect service production by acting on productivities. I developed therefore ways of modelling the time-dependent component of the different services in CRAFTY. The production of a service by an agent in a given year is based on the Cobb Douglas function, adapted to incorporate a time component (Eq. 1).

$$p_s = o_{s,t} \prod_c \left(c_i + \Delta c_{i,t}\right)^{\lambda_c} \tag{1}$$

Production (p_s) of a service s within a cell is the product for all capitals (c) of the optimal production that an agent type would be able to achieve in a given year (o_{st}), and the unit-less (i.e. [0-1]) cell capitals (c_i) plus annual climate-induced change in cell capitals (Δc_{it}), weighted by the capital sensitivities of that agent type (λ_c) (Table B.5). To reflect individual variability, optimal production o_{st} is uniformly randomly drawn from [$0.95\overline{o_{s,t}}$, $1.05\overline{o_{s,t}}$], and capital sensitivity levels from [$\overline{\lambda_c} - 0.1, \overline{\lambda_c} + 0.1$]. Production calculations for each service are described below.

<u>Timber</u>

For timber production, o_{st} is given by a forest owner type-specific function that determines timber growth given forest age. The ProdMod model (Eko 1985) was used to generate timber growth curves for each owner type given their management preferences. Given passive owners' generalised lack of primary objectives for forestry, I assumed them to inherit forest land, and therefore only enabled them to take over the forest and associated optimal production function of other owner types managing forests with the same tree species. Hence, optimal production functions were not calculated for passive owners. Table B.5 shows parameter values used in ProdMod that differed for each owner type. See Appendix B.4.1 for further detail on optimal timber production function calculation.

<u>Carbon sequestration</u>

Due to the difficulty of calculating soil carbon levels in interaction with forest productivities, only above-ground sequestered carbon (excluding the stump) was calculated. Optimal production functions of above ground carbon were also calculated using ProdMod outputs (Appendix B.4.2).

Table B.5 Land owner type production, felling age and competitiveness (scenario-independent) parameters. Number of stems planted per ha for each forest type, site index, number of thinnings implemented, age at each thinning per forest type, and percentage removed per thinning are parameters given to ProdMod to calculate the (age-dependent) optimal timber production and (above ground) carbon sequestration functions. These functions and remaining parameters in this table are CRAFTY-Sweden inputs. Yearly o_s illustrates yearly optimal farmer production. Productivity and infrastructure sensitivities (λ) are given per service. Felling age means (μ) and standard deviations (σ) represent number of years past minimum felling age (m.f.a.) of a forest, given as the m.f.a. range dependent on site quality

	Service Production								***Felling Age***		***Competitiveness***
Land Owner Type	**No. Stems/ha**	**Site Index cm**	**No. Thinnings**	**Age at each Thinning**	**% Removed per Thinning**	**Yearly o_s (tonnes)**	**λ Productiv.**	**λ Infrastr.**	**m.f.a. Range**	**μ, σ Years past m.f.a.**	**Probability Giving-up**
Productionist Pine	2500	280	3	24, 39, 54	25, 20, 20	-	0.8[a], 1[e], 0.06[f]	0.2[a], 1[g]	65-100	12, 10	0.05
Productionist Spruce	2600	360	3	23, 38, 53	25, 20, 20	-	0.8[b], 1[e], 0.06[f]	0.2[b], 1[g]	45-95	10, 8	0.05
Productionist Pine-Spruce	1250, 1300	280, 360	3	24/23, 39/38, 54/53	25, 20, 20	-	0.8[a,b], 1[e], 0.06[f]	0.2[a,b], 1[g]	45-95	10, 8	0.05
Productionist Boreal Br.	2200	320	3	15, 30, 45	25, 20, 20	-	0.8[c], 1[e], 0.06[f]	0.2[c], 1[g]	40-60	9, 7	0.05
Multi-objective Pine-Spruce	1150, 1250	280, 360	2	24/23, 39/38	30, 25	-	0.85[a,b], 1[e], 0.06[f]	0.1[a,b], 0.8[g]	45-95	15, 12	0.05
Multi-objective Pine-Boreal Br.	1840, 420	280, 320	2	24/20, 39/35	25/50, 20/25	-	0.85[a,c], 1[e], 0.06[f]	0.1[a,c], 0.8[g]	65-100	10, 8	0.05
Multi-objective Spruce-Boreal Br.	2000, 420	360, 320	2	23/20, 38/35	25/45, 20/25	-	0.85[b,c], 1[e], 0.06[f]	0.1[b,c], 0.8[g]	45-95	10, 8	0.05
Multi-objective Boreal Br.	2100	320	2	15, 30	30, 25	-	0.85[c], 1[e], 0.06[f]	0.1[c], 0.8[g]	40-60	15, 12	0.05
Recreationalist Pine-Spruce	1100, 1100	280, 360	3	24/23, 39/38, 54/53	25, 20, 20	-	0.9[a,b], 1[e], 0.06[f]	0.3[a,b], 0.6[g]	45-95	80, 14	0.05
Recreationalist Boreal Br.	2000	320	3	15, 30, 45	25, 20, 20	-	0.9[c], 1[e], 0.06[f]	0.3[c], 0.6[g]	40-60	100, 14	0.05

Recreationalist Nemoral Br.	1250, 1250	350, 300	3	25/22, 40/37, 55/52	25, 20, 20	-	0.9[d], 1[e], 0.06[f]	0.3[d], 0.6[g]	110-150	60, 14	0.05
Conservationist Boreal Br.	2100	320	1	15	35	-	0.9[c], 1[e], 0.06[f]	0.3[c], 0.8[g]	40-60	100, 14	0.05
Conservationist Nemoral Br.	1250, 1250	350, 300	1	25/22	35	-	0.9[d], 1[e], 0.06[f]	0.3[d], 0.8[g]	110-150	60, 14	0.05
Passive Pine-Boreal Br.	-	-	-	-	-	-	0.9[a,c], 1[e], 0.06[f]	0.1[a,c], 1[g]	65-100	25, 17	0.05
Passive Spruce-Boreal Br.	-	-	-	-	-	-	0.9[b,c], 1[e], 0.06[f]	0.1[b,c], 1[g]	45-95	25, 17	0.05
Passive Boreal Br.	-	-	-	-	-	-	0.9[c], 1[e], 0.06[f]	0.1[c], 1[g]	40-60	15, 10	0.05
Passive Nemoral Br.	-	-	-	-	-	-	0.9[d], 1[e], 0.06[f]	0.1[d], 1[g]	110-150	10, 10	0.05
Commercial Cereal	-	-	-	-	-	201	0.8[h]	0.5[h]	-	-	0.2
Non-commercial Cereal	-	-	-	-	-	121	0.5[h]	0.3[g,h]	-	-	0.2
Commercial Livestock	-	-	-	-	-	324	0.6[i]	0.5[i]	-	-	0.2
Non-commercial Livestock	-	-	-	-	-	193	0.3[i]	0.2[g,i]	-	-	0.2

Biodiversity

The calculation of optimal forest biodiversity production considered forest age (Duncker et al. 2012b; Koskela et al. 2007; Marchetti 2004), using the generation of coarse woody debris with age as a proxy (e.g. Berg et al. 1994; Jonsell et al. 1998; Siitonen 2001), tree diversity (Gamfeldt et al. 2013; Marchetti 2004) and management practices undertaken by each owner type (e.g. woody debris removal), which have an influence on biodiversity (Chapter 2; Duncker et al. 2012a; Duncker et al. 2012b). I chose these forest attributes as indicators of biodiversity because of the availability of baseline data and the possibility of updating the data during model simulations. Finally, I considered the effect of forest productivity on biodiversity, specifically on coarse woody debris (Sturtevant et al. 1997), by assigning sensitivities to timber productivities. For further details of the calculation of optimal biodiversity production functions see Appendix B.4.3.

Recreation

Recreational value in Scandinavia is largely determined by the age of a forest, but also by forest management practices, accessibility and, to a lesser extent, by the types of tree species present (i.e. conifer vs broadleaf, and monoculture vs mixed) (Edwards et al 2012). See Appendix B.4.4 for further detail on optimal recreation function calculation.

Cereal and meat

Given baseline maps with available capitals and commercial cereal, non-commercial cereal, commercial livestock and non-commercial livestock agent locations (see section 2.1.6), their o_s and λ_c were adjusted until total cereal and meat production equalled the total production in Sweden reported by the FAO (2015) for 2010. The production of non-commercial agents was set at 0.6 times that of the commercial agents to reflect approximate differences in production potentials across equivalent classes in Van Asselen and Verburg (2013).

Timber within a cell is harvested and all service provision is set to zero when a forest is clear-felled. The forest in a cell is clear-felled when it reaches an age that depends on site quality (i.e. productivity) (Lagergren et al. 2012) and owner objectives. In Sweden, the stand age at felling is regulated in law for pine and spruce to guarantee that the production potential is utilised (Kunskap Direkt 2015), and for beech, birch and oak recommended rotation periods exist (Löf et al. 2009; Rytter et al. 2008). Hence, lowest minimum felling age was assigned to the highest productivity values, while highest minimum felling age corresponded to the lowest productivity values (Table B.5). Also, each owner type was assigned a Gaussian distribution of the planned felling age (above minimum felling age) (Table B.5). This distribution was defined as being within the recommended rotation periods for all owner types except for recreationalists, conservationists and passive owners managing broadleaf forests. As these latter groups are not primarily interested in timber production (Chapter 2), they were assigned felling age distributions beyond the recommended rotation period. Felling age is determined at the time that an agent is allocated to a cell by randomly drawing a number (i.e. forest age) from within the agent type's distribution. Upon felling, timber is harvested and carbon that was being sequestered in standing timber is removed from the national pool.

Population, Services, Demand and Benefit. I assume the presence of a population that has a certain level of demand for services D. This represents the needs of the population for consumables such as food and timber, and less tangible demands such as those for biodiversity or recreation (excluding those demands which are fulfilled by imports). The difference between the supply and the demand of ecosystem services is the residual (or unmet) demand, R. The marginal benefit of production (i.e., the benefit attributed to the production of one additional unit of a service) is a function of this residual demand:

(1) $$m_s = u_s(r_s);$$

where m_s is the marginal benefit for service s, u_s is a linear function that describes the benefit of production of service s and r_s is the residual demand for service *s*. As *u(r)* can take negative values, overproduction is actively penalised. For a given bundle of service provision (typically that provided by an agent leveraging a cell), the competitiveness (or benefit) is given by:

(2) $$U_S = \sum_S p_S m_s ;$$

Appendix C

The following document gives supplementary details on the CRAFTY-Sweden model and the methodologies behind its development. The following sections are presented:

C.1 Land owner types - Typology validation

Methods

The validation of the theoretical forest owner typology was done through a comparison with an empirical typology. The empirical typology was based on survey data from a questionnaire distributed among a randomized sample of 3000 Swedish non-industrial private forest owners, which had a response rate of 32% (Vulturius et al. in review). The statistical software R (R Development Core Team 2008) was used to perform a cluster analysis using Ward's method (Ward 1963), in which 91% (i.e. 872 owners) of the respondents (i.e. those who answered all questions used in the cluster analysis) were included.

Ward's method is an agglomerative hierarchal cluster analysis. Hierarchal clustering is an attempt to optimize, stepwise a subdivision (divisive) or synthesis (agglomerative) of data. In the case of agglomerative hierarchal clustering, all data points (n) are initially viewed as individual clusters and the stepwise optimization clusters the data into one group containing all points. At each step, individuals or clusters of individuals are merged together with the most similar individual or group.

A new data frame was created containing only those variables to be considered in the cluster analysis. These were the most similar variables, available from the survey, to those that were found to characterize forest owners in the theoretical supranational typology. These variables included a number of ownership objectives (timber production, tax planning, biofuels, return on investment, income, recreation, berry picking, aesthetic value, biodiversity, environmental protection, water protection, tradition, hunting), and socio-economic attributes (gender, income dependency, property size, experience as forest owners, location of residence).A dissimilarity matrix was created, using "cluster" package, “daisy” command with metric “gower”. The cluster analysis was performed with the Ward’s method using the "agnes" command.

A Chi-square test was applied to test for independency between clusters and different variables. This determined how important the different variables were for the formation of the clusters.

As resulting clusters can be further aggregated or subdivided into sub-clusters depending on the level of aggregation, I looked at two cluster groups (one of which emerged from the other) with a number of clusters close to the number of owner functional types in the theoretical typology. The first group contained four clusters, while the following group contained eight clusters. These clusters were characterized on the basis of agent objectives and socio-economic attributes. While the attributes, personal education and possession of a forest management plan, were not included in the cluster analysis, they were included in the characterization of clusters. Subsequently, clusters were labeled according to their most prominent traits. This resulted in an empirical typology comparable to the theoretical typology. As the empirical typology was only representative for non-industrial private forest owners, a comparison was made with this in mind (e.g. industrial productionists from the theoretical typology would not be included in the comparison).

To have a systematic way of comparing the two typologies, I tabulated the importance of objectives and the socio-economic attributes of the different clusters and sub-clusters in a similar table to the one used for the supranational typology developed in Chapter 2. I used cluster analysis results, which show proportions of respondents who valued an objective to a certain degree along a five point Likert scale (i.e. from not important to very important), to determine which objectives were primary, or secondary, or neither. If the mean value along the scale (0-5) was >4, I considered the objective to be primary. If the value was 3-4, the objective was secondary. While the objectives included in the cluster analysis were in some cases not the same as in the supranational typology, several objectives in the analysis could be grouped under one objective from the typology. In this way, timber production, tax planning, biofuels, return on investment, and income corresponded to profit-making; berry picking and recreation corresponded to personal enjoyment; and environmental protection and water protection corresponded to environmental quality. If one of the ‘sub-objectives’ (e.g. timber production) in each of these groups was a primary or secondary objective, I considered the (broader) objective to be primary or secondary. Socio-economic attributes included in the cluster analysis almost matched one to one the attributes in the supranational typology with the exception of age and property acquisition, which were absent from the analysis.

Results

Cluster characteristics are given in Table B.1. Distinct owner groups according to their objectives and socio-economic attributes can be observed both in the four cluster and eight cluster groups. In the four cluster group two clusters (labeled c1 and c2) can be distinguished with profit-making as their primary objective, and with several other secondary objectives. These resemble substantially the non-industrial productionists and multi-objective owners found in the supranational typology. By contrast, the two other clusters (labeled c3 and c4) show low interest in profit-making, while other objectives are important such as recreational and environmental aspects. In the absence of very important objectives, these clusters are both to a large extent comparable with recreationalists, conservationists and passive owners from the supranational typology.

Cluster c1 was subdivided into clusters c1.1 and c1.2. While the objectives of c1.1 seem to align with those of a multi-objective owner type, its socio-demographic attributes align closely with those of multi-objective and productionist owners. Cluster c1.2 however, having a broad range of objectives with a focus on profit-making and recreational aspects, resembles the multi-objective type.

Cluster c2 breaks down into clusters c2.1, c2.2, and c2.3. Cluster c2.1 aligns very closely with both the non-industrial productionist and the for-profit multi-objective type, while c2.2 resembles the non-profit, multi-objective and the conservationist types. C2.3, with several primary objectives, aligns well with the multi-objective owner type.

Cluster c3 was subdivided into clusters c3.1 and c3.2. While c3.1 aligns well with the recreationalist type, c3.2, with owners who give little value to all objectives, it has the characteristics of the passive owner type. Finally, cluster c4, which did not subdivide (at the same level as with the other clusters), resembles recreationalists the most, even though it does not have personal enjoyment as a primary objective.

Table C.1 a) Clusters of non-industrial private forest owners, which were compared to b) forest owner functional types from the typology described in Chapter 2. Following the characterisation from Chapter 2, owner clusters were featured through their primary (✓✓) or secondary (✓) objectives and socio-economic attributes. Income dependency and location of residence (i.e. residents (R) vs absentees (A)) can be Low (L), Medium (M), and High (H). Educational level, forestry knowledge and property size are categorised for each cluster in relative terms (Lower, Medium (Med.), Higher) with respect to the other clusters. Cluster and owner functional types in bold represent the first level of a hierarchy that branch into one or two consecutive levels respectively, where owner groups not in bold can be found

a)

	C1	C1.1	C1.2	**C2**	C2.1	C2.2	C2.3	**C3**	C3.1	C3.2	**C4**
Profit-making	✓✓	✓	✓✓	✓✓	✓✓	✓	✓✓				
Private Consumption											
Personal Enjoyment	✓	✓	✓✓	✓	✓	✓	✓	✓	✓✓		✓
Public Recreation											
Aesthetics	✓	✓	✓✓	✓	✓	✓✓	✓✓	✓	✓ (3.91)		✓
Nature Conservation	✓		✓	✓		✓ (3.95)	✓	✓	✓		✓
Environmental Quality	✓	✓ (3.05)	✓ (3.88)	✓	✓	✓ (3.98)	✓✓	✓	✓		✓
Cultural Conservation	✓	✓	✓	✓	✓	✓	✓✓ (4.00)				✓
Hunting	✓	✓	✓	✓	✓ (3.09)	✓	✓		(2.95)		
Privacy											
Age											
Educational Level [1]	Lower	Lower	Lower	Lower	Lower	Med.	Higher	Med.	Med.	Lower	Higher
Forestry Knowledge	Higher	Med.	Lower	Higher	Higher	Med.	Higher	Lower	Lower	Med.	Lower
Gender (F/M)	0.23	0.01	0.85	0.23	0.00	0.64	0.69	0.25	0.27	0.16	0.96
Income Dependency	M	L	M-H	M	M	L	M-H	L	L	L	L

Property Size	Larger	Larger	Larger	Larger	Larger	Med.	Much Larger	Smaller	Smaller	Smaller	Larger
Location of Residence	R: H A: L	R: H A: L	R: H A: L	R: H A: L	R: H A: L	R: H A: L	R: M A: M	R: H A: L	R: H A: L	R: H A: L	R: L A: H
Property Acquisition											
Forest Mgmt. Plan (% holders) [1]	58	55	62	72	73	66	81	27	27	26	67

[1] Variable not included in the cluster analysis

b)

	PROFIT-ORIENTED	PRODUCTIONIST	INDUSTRIAL PRODUCTIONIST	NON-INDUSTRIAL PRODUCTIONIST	FOR-PROFIT RECREATIONIST	**MULTI-OBJECTIVE**	FOR-PROFIT MULTI-OBJECTIVE	NON-PROFIT MULTI-OBJECTIVE	**RECREATIONALIST**	**CONSERVATIONIST**	SPECIES CONSERVATIONIST	ECOSYSTEM CONSERVATIONIST	**PASSIVE**
Objectives													
Profit-making	✓✓	✓✓	✓✓	✓✓	✓✓	✓✓	✓✓	✓		✓			✓
Private Consumption						✓	✓	✓	✓	✓			
Personal Enjoyment						✓	✓	✓	✓✓	✓			
Public Recreation					✓✓				✓				
Aesthetics	✓	✓		✓	✓	✓✓	✓	✓✓	✓✓	✓✓			✓
Nature Conservation	✓	✓		✓		✓✓	✓	✓✓	✓	✓✓			✓
Environmental Quality	✓	✓		✓		✓✓	✓	✓✓	✓	✓✓			✓
Cultural Conservation	✓	✓		✓		✓✓	✓	✓✓	✓	✓			
Hunting	✓	✓		✓	✓	✓✓	✓	✓✓	✓	✓			
Privacy						✓	✓	✓	✓	✓			

Attributes													
Age	✓					✓			✓	✓			
Educational Level		Lower ✓				Lower ✓			Higher ✓	Higher ✓			
Forestry Knowledge		Med. ✓				Higher ✓							Lower ✓
Gender	✓					✓			✓	✓			✓
Income Dependency		L, M, H ✓✓				L, M, H ✓✓	L, M, H ✓✓	L, M ✓✓	L, M ✓✓	L ✓✓			L, M ✓✓
Property Size			Much larger ✓✓	Larger ✓✓		Larger ✓✓	Larger ✓✓	Med. ✓✓	Smaller ✓✓	Smaller ✓✓			Smaller ✓✓
Location of Residence		R: H A: L ✓✓				R: H A: L ✓✓			R: L A: H ✓✓	R: L A: H ✓✓			R: L A: H ✓
Property Acquisition	✓					✓				✓			
Forest Mgmt. Plan		✓				✓				✓			✓
Mgmt. Prefs.													
Management Intensity			Intensive, High	Intensive, High	Low, Passive		Medium	Low	Low, Passive		Low, Passive	Low, Passive	Passive

Discussion

Forest owner types in Sweden resemble closely those observed at the supranational level. Most owner types identified in the theoretical typology could be identified among the clusters of non-industrial private forest owners. As industrial and public owners were not surveyed, their empirical categorization was not carried out, and they were therefore not included in the empirical typology. While validation for the Swedish forestry system could not be done using the same method, it is axiomatic within Sweden that the objectives of industrial productionists are for timber production (i.e. primarily economically orientated). Additionally, because Swedish public authorities in forestry matters support a balance between environmental and economic objectives, I assumed in the simulation that public owners correspond to the multi-objective owner type.

Hence, I validated a supra-national theoretical forest owner typology by developing an empirical forest owner typology using survey data relevant to the study area, and comparing them. I can confirm therefore that the theoretical typology is applicable as baseline information in the parameterization and simulation of forest owner decision making in the land-use context.

C.2 Capitals

Capitals that agents use in service production are productivities for pine, spruce, boreal broadleaf, and nemoral broadleaf forests, grassland productivity (for cereal and meat production), and transportation infrastructure. Baseline productivities for the different forest types were calculated by Dr. Mats Lindeskog (Lund University) according to equations and values from (Hägglund and Lundmark 1987) and (Johansson et al. 2013), using tree species, mean forest height, and total wood volume data from SLU Forest Map (SLU 2010) for the year 2010 at 20x20m resolution for Sweden. Productivity values were calculated at a 1km^2 resolution by averaging values from 400m^2 cells. Baseline grass productivities were obtained by extracting the data for the year 2010 from the LPJ-GUESS grass NPP projections from the EC-EARTH model, RCP 4.5 (see section A.7).

Given the relative importance of transport infrastructure in land-use activities, I generated an infrastructure indicator that indicated proximity to transport networks and central markets. To calculate proximity to transport networks, I used data on the locations throughout Sweden of roads (UNECE 2015b), railways (UNECE 2015a), and waterways (EEA 2015), and calculated the Euclidian distance to each of them from every pixel in Sweden at a 1km^2 resolution. I then normalised the distance values for the three resulting layers so that the largest distances would result in the lowest proximity index values, using the equation:

$PI = 1 - D/MaxD$

where *PI* is the proximity index, *D* is the distance to the nearest road, railway or waterway, and *MaxD* is the maximum distance to these calculated for Sweden. Assuming equal importance of the three modes of transport, I then summed the values for each pixel and divided the result by three to obtain an overall index for proximity to transport networks.

To calculate the proximity to central markets I used data on accessibility (i.e. travel times) to the nearest town with a population greater than 50,000 in the year 2000 using land (road/off road) or water (navigable river, lake and ocean) based travel (Nelson 2008). I normalised these data using the above equation, where *D* was the distance to the nearest town. Subsequently, normalised values of proximity to central markets and proximity to transport networks (assuming equal importance of both) were added and divided by two to achieve the final transport infrastructure index.

C.3 Baseline land-use and land owner distribution

<u>Land-use map</u>

I used CORINE land-use data for 2006 at a resolution of 100 m^2 to determine the location of the more distinct land-use types (e.g. agriculture, forest, urban), and SLU Forest Map data (SLU 2010) (which included percentage forest cover, percentage of pine, spruce, boreal broadleaf, and nemoral broadleaf forest volume out of total forest volume, and mean forest age) for 2010 at a resolution of 1 km^2 to determine the type of forest corresponding to the cells identified as forest. Since forest cover in Sweden did not change significantly between 2006 and 2010 (Swedish Environmental Protection Agency 2015), I used CORINE data to identify forest cover locations, and determined the type of forest within each forest pixel using the SLU Forest Map data.

To do this, I first reclassified the land-use categories in the CORINE dataset to: artificial surfaces, agriculture, forest, semi-natural areas, open spaces, wetlands and water bodies. I then aggregated the 100 m^2 cells into 1 km^2 cells using a majority algorithm, which determines the new value of the cell based on the most popular values within the filter window.

To establish forest type locations, I first multiplied percentage forest cover values with species fractions, to obtain the proportion of each forest type in each cell. I then determined forest types based on these proportions. The following rules were defined to determine the forest type to which a pixel would belong (forest type names refer to the proportion of forest types in a cell):

a. Non-forest: Pine + Spruce + Boreal Broadleaf + Nemoral Broadleaf < 50%
b. Pine: Pine ≥ 60%
c. Spruce: Spruce ≥ 60%
d. Boreal Broadleaf: Boreal Broadleaf ≥ 60%
e. Nemoral Broadleaf: Nemoral Broadleaf ≥ 60%, or Nemoral Broadleaf > all others
f. Pine-Spruce: Pine 60-40% and Spruce 60-40%, or Pine and Spruce > Boreal Broadleaf and Nemoral Broadleaf
g. Pine-Boreal Broadleaf (managed by passive owners): Pine 60-40% and Boreal Broadleaf 60-30%, or Pine and (Boreal Broadleaf ≥ 30%) > Spruce and Nemoral Broadleaf
h. Pine-Boreal Broadleaf (managed by multi-objective owners): Pine 60-40% and Boreal Broadleaf 30-20%, or Pine and (Boreal Broadleaf < 30%) > Spruce and Nemoral Broadleaf
i. Spruce-Boreal Broadleaf (managed by passive owners): Spruce 60-40% and Boreal Broadleaf 60-30%, or Spruce and (Boreal Broadleaf ≥ 30%) > Pine and Nemoral Broadleaf
j. Spruce-Boreal Broadleaf (managed by multi-objective owners): Spruce 60-40% and Boreal Broadleaf 30-20%, or Spruce and (Boreal Broadleaf < 30%) > Pine and Nemoral Broadleaf

The proportions for forest types f-j were established based on the proportions planted by forest owner types in ProdMod (see section C.4) based on owner type preferences.

Protected areas were then superimposed on the resulting forest map, to transform cells falling within a protected area to protected land. Non-productive areas, are also protected and cannot be managed for production (Swedish Forest Agency 2014c). These forests are on the least productive forest land. I determined the location of non-productive forests by:

1. Assigning to each cell covered by forest (from the forest type dataset), excluding protected areas, the productivity value corresponding to the forest type with the highest productivity for that cell. The type with the highest productivity was chosen to ensure that non-productive forest areas was selected by considering the potential of every site to grow the most suitable forest type.

2. The Swedish Forest Agency (SFA) Statistics provide the proportion of forest area in each county that is productive and non-productive, so I selected for each county the corresponding proportion of forest cells with the lowest productivity values and labelled them as non-productive.

The resulting forest cover mapped (derived from SLU Forest Map data) covered 71.9% of Sweden, while forest cover from the Corine data was 63%. Therefore, as SLU forest statistics (SLU 2015) report Swedish forest cover to be 69%, I used the forest cover layer to identify forested cells, and superimposed these onto CORINE data. Cells for which the CORINE forest cells did not overlap with our forest cells were converted into unmanaged land. Fig. A.1 shows the final land-use map.

Forest age was attributed according to the mean forest age value in the pixel as identified in the SLU Forest Map.

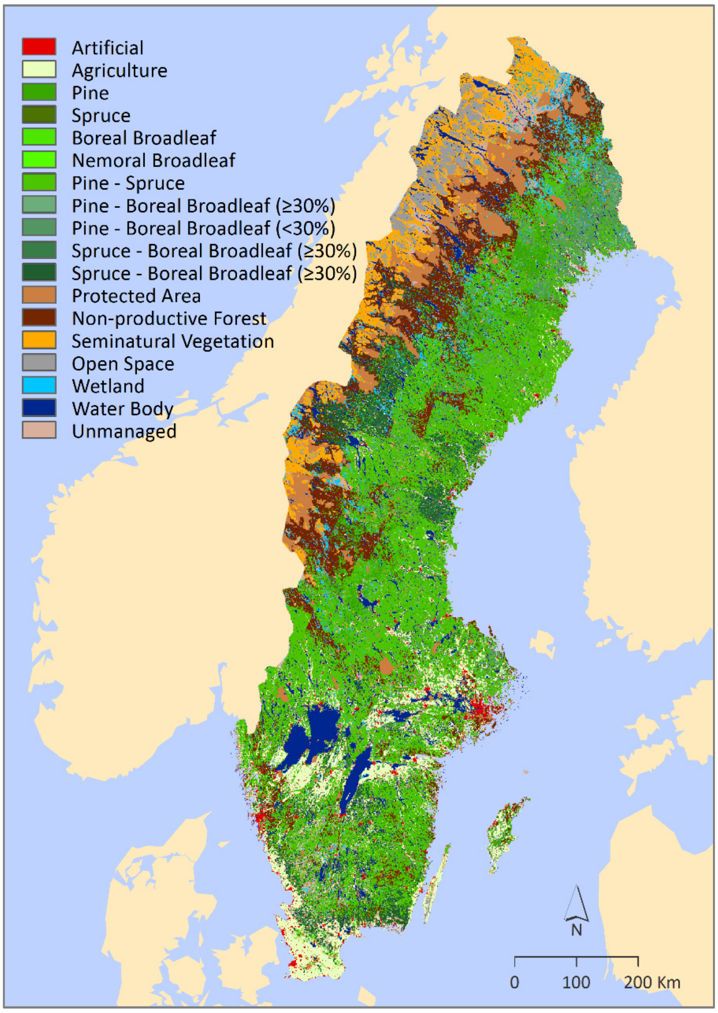


Fig C.1 Land use map generated for Sweden for the year 2010

Agent locations

Since non-productive forest cannot be managed for production, it was combined into one class with protected areas. Agricultural land and (some) semi-natural vegetation were assigned to farmer agents according to the land-use intensity within those areas. I identified five intensity classes according to the Institute of Environmental Studies (2015): extensive arable, moderately intensive arable, very intensive arable, extensive grassland, and intensive grassland. Agricultural land was distributed among the five categories, while extensive grassland was located on semi-natural vegetation (i.e. pasture) where they intersected. Extensive arable land was assigned to non-commercial cereal agents, while moderately and very intensive arable was allocated to commercial cereal agents. This resulted in 4.9% of arable land being allocated to extensive cereal agents, which closely resembles figures reported by the Swedish Board of Agriculture (2009) of 5.0% of the land dedicated to cereal production being used for organic production. Extensive and intensive grassland were allocated to non-commercial and commercial livestock agents respectively.

The remaining semi-natural vegetation, 'unmanaged' land, and wetlands were left unmanaged. Protected areas, non-productive forests, open spaces with little or no vegetation, artificial surfaces, and water bodies were not assigned to any agent, as these land covers are not expected to undergo any important changes at the national scale.

Forest types in productive forest land were allocated to the different owner types for which production functions had been generated. This was done using two different datasets with information about types of forest owners present within the different Swedish counties. These were: a) the area of productive forest land by county and ownership classes for 2010, obtained from SFA Statistics, which classified ownership into state, state owned companies, other public owners, private sector companies, individual owners, and other private owners; and b) the proportion of owners in each county belonging to each cluster obtained from the survey data cluster analysis. Owners who had participated in the survey were considered to correspond to individual owners in the SFA data.

As all owners surveyed owned productive forest, their replies, and consequently the cluster profiles, were considered appropriate to determine the proportion of productive forest land owned by each cluster. Since I was able to relate clusters to owner types identified in the theoretical typology, I could use cluster proportions per county to determine the proportions of owner types within non-industrial private forest owners.

The state, state owned companies, and other public owners, because they promote sustainable forestry that balances its economic, environmental and social aspects (see Chapter 5), were considered to be multi-objective owners. Private sector companies were identified as productionists. As I found no information about other private owners that could help to decide on one or more owner types to define this group, I divided the proportion of land owned by them equally among the five possible owner types.

Hence, I could calculate the proportion of owners within each SFA owner class per county and subsequently calculate the proportion of owner types in each county. In the case of SFA individual owners, I calculated their distribution among the different owner types by multiplying the proportion of this owner class in each county by the proportions of each owner type derived from the cluster analysis per county.

I then counted the number of cells of productive forest and of each forest type per county, and calculated the number of productive forest cells per county corresponding to each owner type according to their proportions. Next, owner types were assigned to forest types. As pine and spruce could only be assigned to productionists, pine-boreal broadleaf (<30%) and spruce-boreal broadleaf (<30%) could only be assigned to multi-objective owners, and pine-boreal broadleaf (≥30%) and spruce-boreal broadleaf (≥30%) could only be assigned to passive owners, the number of cells of these forest types were allocated to the corresponding owner types. The remaining number of agents of each owner type were allocated by assigning probabilities of each owner type of managing each forest type in each county. These probabilities were a function of the likelihood of an agent type occurring in a county (i.e. number of agent types per county) and the likelihood of them managing a certain type of forest. The latter was dependent on the number of agent types that a forest type could be assigned to. For instance, if nemoral broadleaf could only be managed by recreationalists, conservationists and passive owners, each agent type would have a 33.33% chance of being assigned to that type of forest.

C.4 Land owner service production

B.4.1 Timber

For timber production, o_{st}, being the optimal production that an agent type is able to produce with optimal capital levels, corresponds to the production (timber volume) for a given year within the growth period of the forest type (i.e. age-dependent production). This value is calculated by CRAFTY using functions of maximum yearly standing volume that were generated after post-processing ProdMod's model output of maximum timber growth. Prodmod is an empirical stand growth and yield model (Eko 1985). ProdMod simulations were run for all other owner types at a latitude of 57°, altitude of 100m, with an understory of herbs and grasses, an initial basal area of 0.5m^2/ha, and breast-height age (i.e. elapsed time since tree height exceeded breast height) of 1 year.

To generate maximum timber production equations I 1) plotted the timber volume data against the corresponding stand age, and 2) adjusted trend lines in MS Excel to the generated curve. Among the possible trend lines I chose the one that fitted the curve best (i.e. with the highest R^2). Possible trend line functions to fit the data were exponential, linear, logarithmic, polynomial, and power. Polynomial equations gave the best fit for all timber growth datasets (Fig. A.2). To achieve the best possible fit, degree two polynomial equations with R^2<0.98 were substituted for degree three polynomial equations if the latter had a better fit to the data. A similar procedure was used to adjust best fitting functions to above-ground carbon growth data.

Capitals used in the production of timber were forest (type) productivities and transportation infrastructure. I attributed infrastructure sensitivity values 0.3-0.5 according to the socio-demographic attributes (i.e. property size) of the owner types, and to their baseline distribution throughout Sweden, so that if an owner concentrated for instance in the south, where highest infrastructure levels occur, they was assumed to be more sensitive to infrastructure.

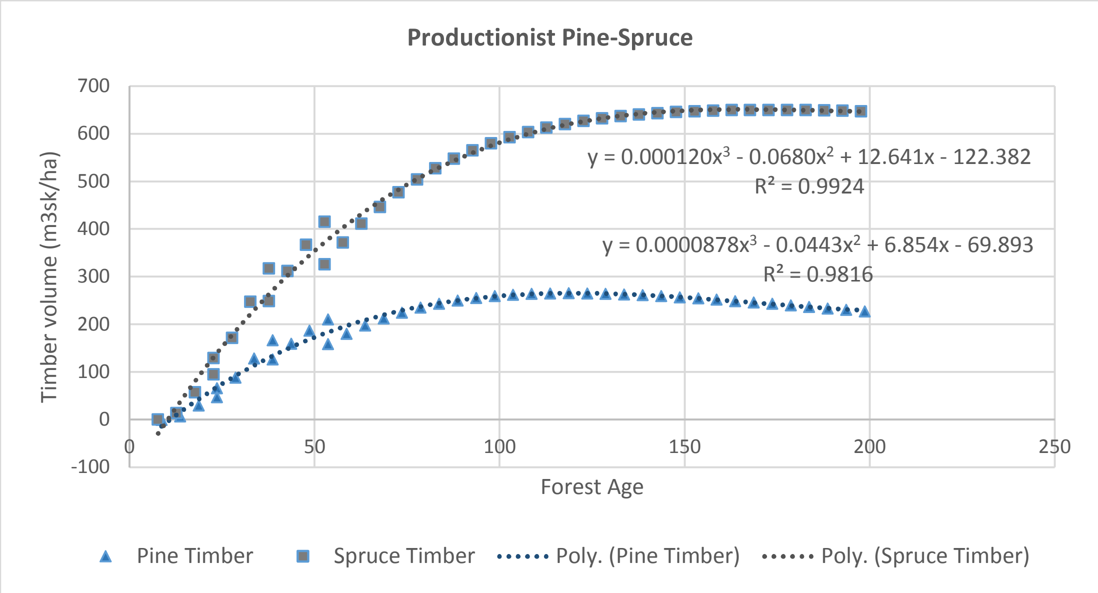


Fig C.2 Example of curves of timber growth through time for a productionist managing pine and spruce (dotted lines), generated from adjusting trendlines to ProdMod timber volume data (triangles and squares). On years when thinning takes place two volume values can be seen (before and after thinning). R^2 values show the goodness of fit of the curves to the data.

C.4.2 Carbon sequestration

For sequestered carbon, I calculated maximum productions functions of above ground carbon (excluding the stump) to be used in the Cobb Douglas equation following a similar procedure as that for timber production. This could be done for all forest types except nemoral broad-leaved forests using above ground biomass (excluding the stump) results from ProdMod simulations, and multiplying them by the carbon fraction of aboveground forest biomass given by the IPCC (2006), which is different for conifers (0.51) and broad-leaved forests (0.48).

Because ProdMod does not generate biomass data for beech and oak, I could not use the simulation results to inform above ground biomass of nemoral broad-leaved forests. Instead, to calculate this biomass, I used ProdMod's five-yearly volume per hectare results (which refer to the entire stem cone above the bark), and added them to estimations of branch volume (down to a diameter of 5cm) per hectare for both beech and oak, which I calculated using the following equations (Hagberg and Matérn 1975):

$$Beech\ branch\ volume\ (m^3) = (0.02080*D^2*H - 0.24212*D*H - 0.0003486*D^2*H^2)*0.001*S$$

$$Oak\ branch\ volume\ (m^3) = (0.02813*D^2*H - 0.3178*D*H - 0.0006658*D^2*H^2)*0.001*S$$

where *D* is the diameter over the bark at breast height (cm), *H* is the height from the ground (m), and *S* is the number of stems per hectare.

I then multiplied the resulting above ground volumes of beech and oak by their corresponding wood densities, 580 and 720 kg dry matter/ m^3 fresh volume respectively (Cienciala et al. 2005; IPCC 2006; Södra 2013) to obtain above ground biomass. Södra (2013) suggests that oak density may range in Sweden from between 690 and 760 kg/m3 (at 15% moisture content). As oak density decreases with increasing growth rate, assuming a linear relationship, I chose a medium density value, given that I am simulating the fastest possible growth (i.e. maximum potential production per unit of time) for nemoral broadleaf forests managed by low intensity forest owner types (i.e. recreationalist, conservationist, and passive). Aggregating the biomass of both species results in nemoral broad-leaved forest biomass.

For all owner functional types managing more than one type of forest (e.g. productionist pine-spruce), the carbon content values of both forest types were aggregated for each time step to obtain the total maximum carbon content that could be generated by the agent type.

Since carbon is incorporated by a tree as it grows in volume, and I confirmed that timber volume production and above ground carbon production are strongly linearly correlated (tested with ProdMod simulation results for different agent types) (i.e. they grow at the same rate with time), I could assume that, just like timber volume production, above ground carbon production could be underpinned by forest type dependent productivities. Hence, I could use forest land productivity with maximum carbon production functions in the Cobb Douglas function.

Additionally, carbon production was recorded as negative when the forest was felled to reflect carbon removal.

C.4.3 Biodiversity

The calculation of optimal forest biodiversity production considered forest age (Duncker et al. 2012b; Koskela et al. 2007; Marchetti 2004), tree diversity (Gamfeldt et al. 2013; Marchetti

2004) and management practices undertaken by each owner type (e.g. woody debris removal), which have an influence on biodiversity (Chapter 2; Duncker et al. 2012a; Duncker et al. 2012b). I chose these three forest attributes as indicators of biodiversity because of the availability of baseline data and the possibility of updating the two capitals during the model simulations.

In a nationwide study in Sweden, Gamfeldt et al. (2013) reported understory plant species richness to be 31% greater in forests with five rather than one tree species. Hence, I estimated the increase in forest biodiversity (measured on a scale of 0-1) with each additional tree species to be (31/4) 7.75%. Five classes resulted from separating forest types according to the number of tree species, with forests with five or more tree species being assigned a value of 1. One of the most important factors for sustaining a rich biodiversity in the boreal forest landscape is the presence of deadwood (e.g. Berg et al. 1994; Jonsell et al. 1998; Siitonen 2001). Hence, I used dead woody debris (i.e. deadwood with a minimum diameter ≥10 cm) as a proxy for biodiversity. I parameterised the effect of age on biodiversity by estimating dead woody debris volume as a logistic function of age using the model of coarse woody debris accumulation from Sturtevant et al. (1997), while I normalised volume between 0 and the maximum volume estimated at the age where the curve saturates (Figure C.3).

Different owner types may have different impacts on biodiversity depending on their management practices (e.g. woody debris removal) (Duncker et al. 2012b). The effect of management was considered for each owner type as a multiplying factor, and was parameterised using the findings of (Duncker et al. 2012a; Duncker et al. 2012b) as guidance.

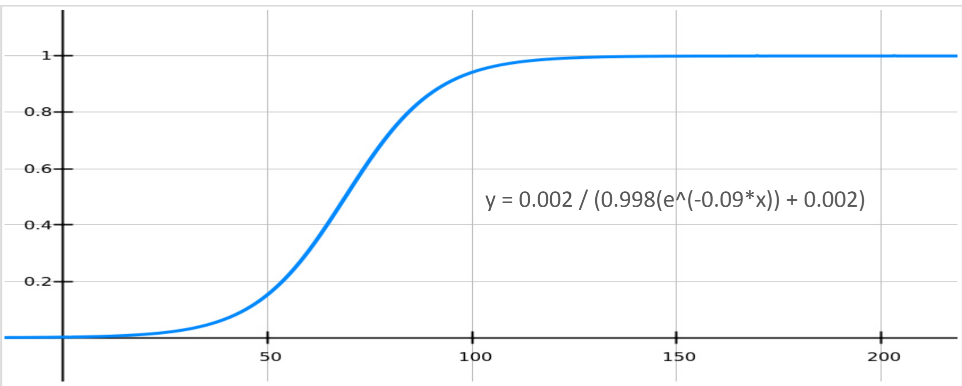


Fig C.3 Equation used to estimate coarse woody debris accumulation in forests at a certain forest age.

The biodiversity production function looked like:

$B = ME*NV_t*TD$

where, *ME* represents the effect of management, NV_t is normalised coarse woody debris volume as a function of time, and *TD* is the effect of tree diversity on understory plant species richness.

Finally, as biodiversity, and specifically coarse woody debris, can be expected to be affected by forest productivity (Sturtevant et al. 1997), I considered forest productivity as a capital affecting this service. Additionally, I attributed a conservative (i.e. low) capital weight of 0.06 to forest productivity. For agents managing more than one forest type (e.g. pine-boreal broadleaf), the productivity of the most abundant forest type under its management was chosen.

C.4.4 Recreation

Recreational value in Scandinavia is largely determined by the phase of development of a forest, but also by forest management practices, and, only to a much smaller degree, by the type of tree species present (i.e. conifer vs broadleaf, and monoculture vs mixed) (Edwards et al. 2012). Additionally, travel time and distance are well known to determine the accessibility of recreational areas. Recreation in forests was therefore assumed to be a function of forest age, tree species type, accessibility (i.e. infrastructure capital), and management practices. To parameterise production weights I used the results from the study by Edwards et al. (2012) to create an equation to calculate the maximum possible production of an agent type with a forest of a certain age:

$R = A*S*M$

where *A* is the score assigned to a forest given its age, as estimated through the function shown in Figure C.4, which was calculated from recreational value scores assigned to phases of forest development (i.e. establishment (0-5 years), young (5-15 years), medium (15-50 years) and adult (50+ years)). For modelling purposes, the result of this function is set to 0 for values where x=0. *S* and *M* are the scores given its tree species combination (i.e. conifer, broad-leafed or mixed) or management strategy respectively (the highest score for each forest attribute being 1). Table C.2 shows S and M scores.

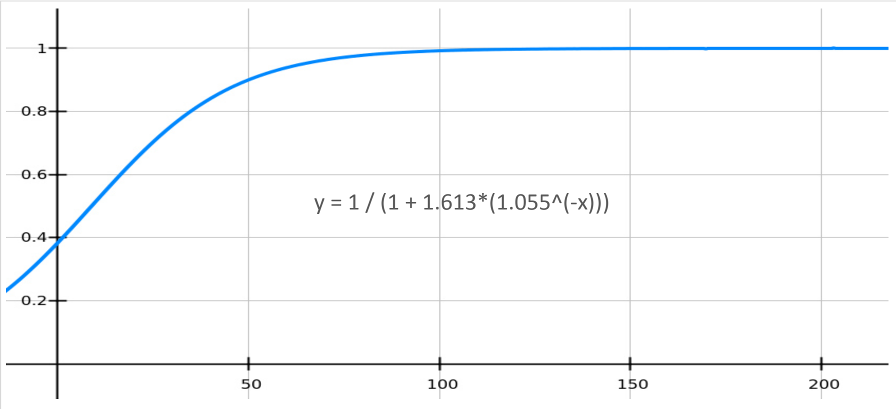


Fig C.4 Equation to estimate recreational value of a forest at a certain forest age.

Table C.2 Scores assigned to forest owner types given their species combination (*S*) and management strategy (*M*) to calculate their provision of recreation

Forest Agent Type	***S***	***M***
Productionist Pine	0.95	0.84
Productionist Spruce	0.95	0.84
Productionist Pine-Spruce	0.95	0.84
Productionist Boreal Br.	0.98	0.84
Multi-objective Pine-Spruce	0.95	0.96
Multi-objective Pine-Boreal Br.	1.00	0.96
Multi-objective Spruce-Boreal Br.	1.00	0.96
Multi-objective Boreal Br.	0.98	0.96
Recreationalist Pine-Spruce	0.95	1.00
Recreationalist Boreal Br.	0.98	1.00
Recreationalist Nemoral Br.	0.98	1.00
Conservationist Boreal Br.	0.98	0.94
Conservationist Nemoral Br.	0.98	0.94
Passive Pine-Boreal Br.	1.00	0.90
Passive Spruce-Boreal Br.	1.00	0.90
Passive Boreal Br.	1.00	0.90
Passive Nemoral Br.	0.98	0.90

As accessibility is primary in enabling recreation, assigned infrastructure capital sensitivities were between 0.6 and 1. Productionists and passive owners were given levels of 1, under the assumption that because recreation is not among their objectives their forests will have little or no recreational value if they are not easily accessible. In contrast, because recreation is recreationalists main objective, they were assumed to still be able to provide this service even if they are further away, and were therefore assigned levels of 0.6. Multi-objective owners and conservationists were assigned 0.8 for having recreation as a secondary objective.

C.5 Forest Felling

The forest can be felled within a defined age distribution for each manager type. In Sweden, the time of felling is regulated by law for pine and spruce. Minimum age at clear felling depends on the site index with some small differences between counties. The site index is a measure of site suitability to grow one or other tree type, so the minimum age of felling can be established in relation to the forest productivity of the cell using linear interpolation of rotation length between the highest and lowest productivities (Lagergren et al. 2012). Highest productivity values can correspond to lowest minimum felling age, while lowest productivity values correspond to highest minimum felling age.

In Sweden, the minimum age for final felling in stands where at least half the volume is made up of pine, at the most fertile sites, is 65 years, and at sites with poorer soils it is 100 years. The minimum age for spruce at most fertile sites is 45 years, and at least fertile sites it is 90 years (Kunskap Direkt 2015). Such regulation does not exist in Sweden for other tree species though. For beech, recommended rotation time is 100-120 years, and 120-180 years for oak (Löf et al. 2009). As these two species make-up the nemoral broadleaf forest type, I defined the recommended rotation times for the type to be the mean of the maximum and minimum ages for the period for each species (i.e. 110-150 years). For birch the recommended rotation time is 40-60 years (Rytter et al. 2008). Having the highest and lowest minimum final felling ages for pine and spruce and recommended felling ages for broadleaf species, I can plot them against the maximum and minimum productivity capital values (i.e. 0-1) present at baseline and subsequently interpolate a linear trend line between the two points. The resulting equations for pine and spruce define the minimum age of felling for each cell, beyond which an agent with an interest in timber production needs to go before felling its forest, while for broadleaf species the equation establishes an approximate age at which the forest should be felled. For recreationalists, conservationists and passive owners managing broadleaf forests, as they are not primarily interested in timber production, I defined felling ages beyond the recommended rotation period.

Agent types with more than one tree species (e.g. productionist pine-spruce) present two different minimum clear felling ages, even though a cell in CRAFTY-Sweden is entirely cleared at the time of final felling. To have only one felling age for each cell, I chose the minimum felling age function corresponding to the forest type that would generate the most timber volume according to ProdMod results, while ensuring that this remained consistent with owner type objectives.

C.6 Climate change impact on productivities

I first calculated yearly timber volume growth (m^3 ha^{-1} yr^{-1}) using LPJ-Guess projections for each forest type for each GCM-RCM ensemble and each RCP. Each data value is given for a 50 km^2 cell, and is an average for all forest patches with all ages represented within the cell. This cell resolution is sufficient because I am interested in the changes in its values associated to climate change, which is a large scale phenomenon that would not cause significant differences at smaller scales (e.g. 1 km^2). To calculate yearly volume growth I applied the equation:

$$G = V_t - V_{t-1} + H \times 10/(0.4 \times 0.5)$$

where V is standing timber volume (m^3sk ha^{-1}, m^3sk being the volume of the entire stem cone including bark) and H is the harvested carbon (kg C m^{-2}). The 10 results from the unit conversion 10000 m2/ha × 1 ton/1000 kg, 0.4 is wood density (tonne dry wood/m3 wood), and 0.5 is the carbon fraction of wood biomass (tonne C/ton dry wood). This calculation of volume growth assumes that the wood density of all species is the same, which is not true. Therefore, to be able to compare the different forest types, volume growth was adjusted by a factor of 4/5 for boreal broadleaf and 2/3 for nemoral broadleaf.

I then calculated a linear equation for each cell for the different forest types with yearly production values, and also for grass using grass net primary productivity (NPP), which provides the yearly increase in the capital attributable to climate change. I did this by creating an equation of change in timber production and grass NPP with time for a past period (1951-2000) and another one for the future (2010-2100). Yearly increase values for each cell where then downscaled to 1 km^2 by dividing the 50 km^2 cells, while maintaining the same values. Because the grid of values did not cover the whole of Sweden due to the original coarse data resolution, I assigned to locations with no growth/ NPP values the values of the nearest neighbours.

I normalised the baseline grassland productivity values between 0 and 1, where 0 is the minimum possible productivity in Sweden and 1 is the maximum productivity. Since the values of forest productivities were not available for the whole of Sweden, I assigned values to locations where they were missing by attributing to them the mean value of the ten nearest neighbours. Because nemoral broadleaf baseline productivity was only available for the southernmost part of the country, I attributed values to locations missing productivities by assuming decreasing south-north (because nemoral forests are adapted to nemoral climate, which occurs in the south of Sweden) and east-west (because mountains cover part of the west of Sweden) gradients, while also attempting to assign new values in the south that are somewhat close to the existing ones. I also normalised the growth and NPP rates of change by the maximum baseline productivity value of the corresponding forest type and of grass. Finally, each year's forest productivity in a cell was calculated using the equation:

$P_t = P_{t0} + (Y * R)$

where P_{t0} is the normalised baseline productivity, Y is the number of the years into the simulation (e.g. if 2012, Y=2), and R is the normalised yearly increase. This may give us in some cases values >1, which would be reasonable because the capital is multiplied by the production weight in the Cobb Douglas function. As the production weight reflects the maximum production that an agent can achieve in the present (i.e. it is not climate sensitive), in the 'future', in a cell with a high productivity value the impact of climate change on the productivity could allow it to go beyond its maximum production capacity in 2010.

C.7 Scenario analysis

Demand projections

Baseline (i.e. for 2010) demands for timber, cereal and meat were assumed to be equal to observed production in 2010. Timber production for the different forest types were calculated using data about annual gross fellings and industrial consumption of roundwood by species (Swedish Forest Agency 2015). Cereal and meat production were sourced directly from (FAO 2015). As no reliable empirical data to establish baseline demands were found for sequestered carbon, biodiversity and recreation, these were assumed to be equal to the modelled aggregate supply in 2010.

To estimate changes in demands through time for the different scenarios for timber, carbon, cereals and meat I used the IIASA SSP data (IIASA 2015). Projections of forest land cover until 2100, modelled under SSPs 1, 3 and 4 in combination with RCP 4.5, were used to calculate decadal rates of change that were applied to the baseline demands for timber and carbon under the assumption that global forest cover area positively correlates with timber and carbon demands. Because the IIASA projections were not calculated for RCP 8.5, I assumed forest cover to follow the linear trend established between RCPs 4.5 and 6.0 for each year, which I extrapolated for RCP 8.5. The same process was used to estimate changes in demands for cereal and meat, although I used IIASA modelled crop and livestock demands instead of forest cover to calculate the rate of change in demands. Changes for biodiversity were estimated following the SSP storylines and with guidance from modelled global future changes in species abundance from UNEP (2007). Rates of change in demands for recreation were assumed to be the same as those for biodiversity.

Additional simulation parameters

At each time step (i.e. year) 1% of available cells can be allocated to agents. Also, a number of potential new agents (equal to 0.6% of cells) search cells each year, and each of these agents search 0.01% of available cells.

Appendix D

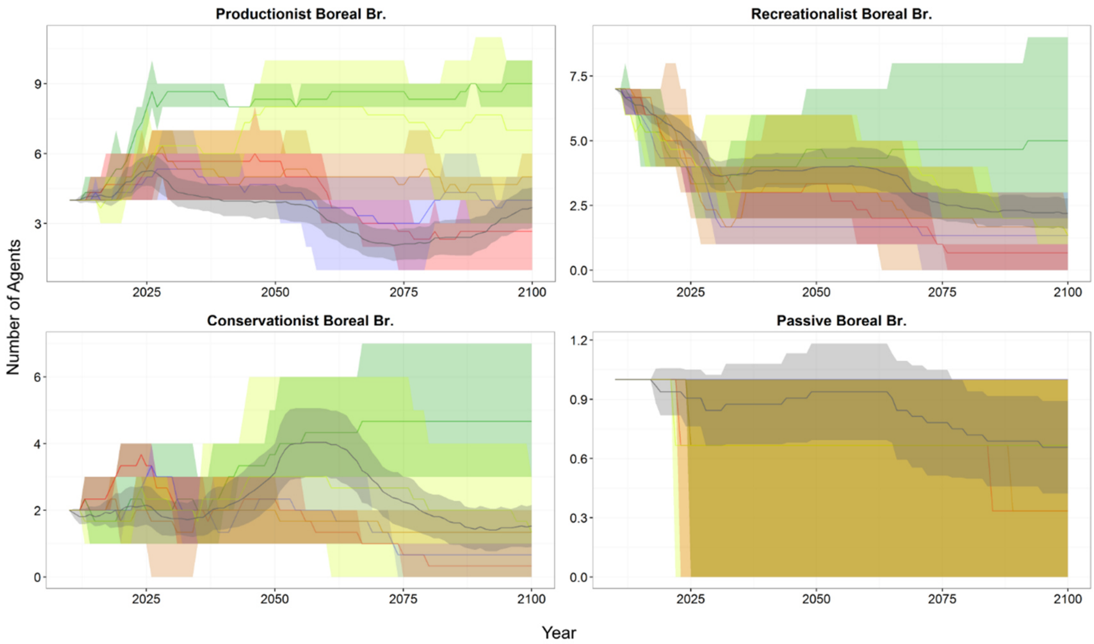


Fig. D.1 Ranges of the number of agents for management strategies implemented by ten agents or fewer through time, given by the results from the three climate models (shaded areas) and their means (solid lines) for the five SSP-RCP combinations, and the Reference scenario (mean of 32 variations of random seed).

Appendix E

Institutional Narratives

<u>Government</u>

The Swedish forest policy of 1993 (i.e. Forestry Act) places equal emphasis on production goals and environmental goals. Policy instruments used before the 1993 Forestry Act - detailed regulation, economic incentives, command and control monitoring, and enforcement - were 'softened' to focus on information and education, advice, extension services and voluntary agreements (Schlyter et al. 2009). Nevertheless, the political aim of the environmental goal as formulated in the Forestry Act is still ambitious, and well above legal requirements of forest owners/ managers. The main points covered by the Act concern forest felling, reforestation, insect damage, nature consideration, cultural heritage and the reindeer husbandry.

The Forestry Act states that thinning must encourage forest development (Swedish Forest Agency 2014c), and timber stocks following thinning must be large enough to make use of the production capacity of the land. Damage to trees and soils must be avoided as much as possible. Regeneration felling (i.e. final felling) must not be carried out until the forest has reached a certain age. For prevailingly coniferous forests, the age varies between 45 and 100 years. Regeneration felling is restricted on forest properties larger than 50 hectares. For these properties, up to half of the land may be made up of finally felled areas and of stands below 20 years old. Additional rules apply to properties larger than 1000 hectares. After felling, new forest must be planted or naturally generated when the land's capacity to produce timber is not fully exploited. Planting or measures for natural regeneration must have been completed by the end of the third year after felling. Unmanaged agricultural land must be reforested within three years of the land falling into disuse. The latter does not, however, apply to land to be protected for its natural features or its cultural heritage.

The Forestry Act also stresses that biological diversity and cultural heritage in forests must be safeguarded, while social aspects must also be considered (Swedish Forest Agency 2014c). Some important issues include not creating excessively large felling areas, leaving non-productive forest land untouched, and leaving protective buffer zones adjoining water, non-productive land, agricultural land and urban areas. Where there is a choice of methods to be used, priority must always be given to the promotion of biological diversity. At the same time, the conservation requirements must not be so far-reaching that forestry activities become substantially more difficult.

At the national level, the Swedish Forest Agency (SFA) and the Swedish Environmental Protection Agency (SEPA) operate under the supervision of the Ministry of Rural affairs and the Ministry of the Environment respectively. These ministries establish annual national goals that the agencies are expected to fulfil. The SFA is responsible for the implementation of forest policy, which entails providing information and advice to forest owners, monitoring rule compliance, administering subsidies, issuing felling permits, establishing small protected areas (i.e. habitat protection areas and nature conservation agreements), and coordinating climate change adaptation sectorally (Appelstrand 2012; Keskitalo et al. 2012; Swedish Forest Agency 2014b). However, according to some scholars, the SFA has a rather unclear role and an increasingly uncertain existence (e.g. Sundström 2005, as cited in Appelstrand 2012). The SEPA complements the roles of the SFA, being the organism responsible for the implementation of environmental policy. Its principal duties involving the forestry sector include the management of National Parks, and national and international follow-up and reporting on climate change adaptation strategies (Keskitalo et al. 2012; Swedish Environmental Protection Agency 2014b). The SEPA also collaborates with the SFA in working areas where environmental matters overlap with forestry such as climate change or forest biodiversity conservation (Swedish Environmental Protection Agency 2014a).

Regionally, 21 county administrative boards coordinate the development of the county across municipalities in line with national goals. They are in charge of the establishment and maintenance of nature reserves, in collaboration with municipalities, and of regional coordination of sustainability and climate change adaptation measures (Angelstam et al. 2011; Appelstrand 2012; Lundstrom et al. 2014; Ulmanen et al. 2012). Regional Forest Agency Offices are in charge of the provision of services offered by SFA to forest owners (advice, permits, subsidies/payments) and monitor their activity at the regional level (Kleinschmit *pers comm.* 2014).

Municipalities hold a planning monopoly at the local level (Keskitalo et al. 2012). These institutions are given the responsibility for local actions on sustainability and on climate change adaptation, while their implementation is dependent on municipal priorities (Keskitalo and Liljenfeldt 2012). Locally, Regional Forest Agency Offices branch into Forest Agency Districts, level at which the SFA monitors forest owner activities to ensure observance of the Forestry Act and offers them their services (Angelstam et al. 2011; Swedish Forest Agency 2014a).

Research Suppliers

The main generators of forestry-related research in Sweden are the Forestry Research Institute of Sweden, the Swedish Meteorological and Hydrological Institute, the Swedish Biodiversity Centre, the Swedish University of Agricultural Sciences, the Royal Swedish Academy of Agriculture and Forestry, and Mistra-SWECIA (Swedish Research Programme on Climate, Impacts and Adaptation) (Ulmanen et al. 2012, André *pers comm*. 2014). These institutions monitor and research the bio-physical and socio-economic environment, and inform and advise government agencies on their findings.

There are also companies and other organizations that provide advisory services to forest owners such as 'Skogsällskapet', an independent forestry and business partner, and the Swedish Rural Economy and Agricultural Societies, as well as other sawmill and wood buying companies (Blennow 2008). As far as providing information and advice goes, these essentially work as a bridge facilitating the information that research organisations generate to forest owners.

Lobbyists

Lobby groups are characterised by institutions that try to influence (lobby) other institutions. Lobbying is used here as a generic term for attempts to influence public decisions (Bjärstig and Keskitalo 2013). The existing conflicts between nature conservation and forestry (Kleinschmit et al. 2012) define two lobby groups with generally diverging values and interests: environmental NGOs and forest owner associations.

The main environmental NGOs currently involved in Swedish forestry are the Swedish Society for Nature Conservation, Friends of the Earth Sweden and WWF-Sweden. Their goal is to promote nature conservation and sustainability, including climate change mitigation and adaptation, and they act at the national and international levels, but they also have a regional and local presence (Jordens Vänner 2014; Naturskyddföreningen 2014; Världsnaturfonden 2014). They do this by providing information and lobbying the SFA and the SEPA. They also take legal action to influence governmental decisions with which they do not agree, such as appealing SFA decisions to provide logging permits, through the courts of law (Vulturius *pers comm.* 2014).

Forest owner associations act on behalf of the interests of their members, at a range of different scales, from the individual forest owner level to international institutions. These organizations often provide services such as certification or logging for their members (Keskitalo and Pettersson 2012). In addition, forest owner associations also engage in lobbying. At the national level, two main associations exist: the Swedish Forest Industries Federation and the Federation of Swedish Family Forest Owners. While members of the former are industrial forest companies, the second is made up of non-industrial private forest owners. The Swedish Forest Industries Federation is involved, along with its member companies, in Swedish and European industrial policy-making (Bjärstig and Keskitalo 2013). They lobby the government through the Ministry of Rural Affairs (mostly) and the Ministry of Environment (Vulturius *pers comm.* 2014). Also, as a member of the Confederation of European Paper Industries, they handle all the work at the European level on behalf of their member companies (Bjärstig and Keskitalo 2013). In turn, the main focus of the Confederation of European Paper Industries' lobbying efforts is the European Commission, while the parliament is becoming increasingly more important. Besides their membership of the Confederation, the Swedish Forest Industries Federation contacts and interacts with the Swedish parliament and the Swedish representation in Brussels when necessary. In turn, ministries, and European institutions in particular, benefit from the information that owner associations provide them. Some large forest companies also complement their efforts as members of these forestry associations with their own lobby activities.

At the regional level, the Federation of Swedish Family Forest Owners branches into regional forest owner associations (Lantbrukarnas Riksförbund 2014). Four regional forest owner associations draw their memberships from different regions in the country, namely Södra (South), Norra Skog (North), Norrskog (Mid-north) and Mellanskog (Mid-south).

Supranational Institutions

The range of forest relevant institutions at a supranational level extends from legally binding international conventions such as the Convention on Biological Diversity or the United Nations (UN) Framework Convention on Climate Change, to non-binding intergovernmental negotiation settings such as the UN Forum on Forests, to private norm setting initiatives (i.e. certification schemes). There are also international knowledge providers, amongst which the Food and Agriculture Organization (FAO) is important. The FAO Committee on Forestry and the FAO/UNECE (UN Economic Commission for Europe) Timber Committee generate information and openly provide advice to nations and others. The Ministerial Conference on the Protection of Forests in Europe (MCPFE), a process now known as Forest Europe, act at the pan- European policy level develop guidelines and instruments (e.g. sustainable forest management) that can be taken up at the national level (Kleinschmit and Edwards 2013). Also, international environmental NGOs, such as WWF or Greenpeace among a long list of others, are involved in the creation of conventions and treaties aiming for international laws and policies that ensure the sustainable management, effective protection, and equitable use of biodiversity and natural resources (Pattberg 2005; Reimann 2006). They also provide advice and engage in lobbying at the European level.

The European Union (EU) establishes its own directives and soft laws that member countries are expected to follow. The EU Forestry Strategy sets a non-binding framework to support sustainable forest management and the multifunctional role of forests. While forestry is not currently a regulated area under the EU, matters relating to forest use and forestry are taken up in a number of other policy areas such as environment, biodiversity, habitat, and water management (European Commission 2014). Sweden has adopted forest relevant EU directives such as the Habitats Directive, the Birds Directive and the Water Framework Directive. While the objectives of an EU directive are binding, its interpretation and subsequent implementation are left to the discretion of each nation (Keskitalo and Pettersson 2012). When it comes to the EU policy process, the European Commission actively tries to network with different kinds of interests groups, which include relevant ministries and forest owner associations, in order to obtain expert knowledge and to involve different stakeholder groups early on in the policy process (Bjärstig and Keskitalo 2013).

Certification has been described as ‘market-bsased regulation’ (Cashore et al. 2004). Sweden is believed to be the state with the largest proportion of certified forests in the world

(Keskitalo and Pettersson 2012). The Forest Stewardship Council (FSC) and the Pan-European Forest Certification Council (PEFC) play a comparably large role in the country (Cashore et al. 2004; Hysing 2009). They both set higher requirements than the law. The FSC holds a broad and demanding set of standards for certification (Meidinger 2006). FSC's organizational structure consists of a General Assembly that comprehends three equally weighted chambers with stakeholders from the environmental, social, and economic domains (Werland 2009). The PEFC, most closely identified with landowners and industry and controlled largely by traditional forest interests, was set up as a response to the establishment of the FSC, which was deemed to be dominated by ecological and social interest groups.

Appendix F

Table F.1 Parameters given at t_0 and subsequent parameters and generated variables at t_1 for one model simulation. Institutions may choose between the behavioural options: lobby (Lob.), subsidise (Sub.), set production quotas (S.Q.), invest in infrastructure (Inf.), or take no action (N.A.). In this example, action priority values above 0.2 (implementation threshold) lead to an action being implemented on the corresponding service.

Time Step	Institutional Type	Service	Service SDD	Service Preferences	Situational Service Priorities	Perceived Effectiveness of Potential Actions				Action Priorities				Implemented Actions			
						Lob.	Sub.	S.Q.	Inf.	Lob.	Sub.	S.Q.	Inf.	Lob.	Sub.	S.Q.	Inf.
t_0	Environmental NGOs	Timber	0.6	0.3		0.5											
		Biodiversity	-0.5	0.9		0.8											
		Recreation	-0.1	0.5		0.6											
	Owner Associations	Timber	0.6	0.9		0.8											
		Biodiversity	-0.5	0.3		0.5											
		Recreation	-0.1	0.4		0.6											
	Government	Timber	0.6	0.9			0.8	-0.8	0.7								
		Biodiversity	-0.5	0.9			0.8	-0.6	0.6								
		Recreation	-0.1	0.6			0.8	-0.6	0.8								
t_1	Environmental NGOs	Timber	0.6	0.3	-0.18	0.5				-0.09				N.A.			
		Biodiversity	-0.5	0.9	0.45	0.8				0.36				Lob.			
		Recreation	-0.1	0.5	0.05	0.6				0.03				N.A.			
	Owner Associations	Timber	0.6	0.9	-0.54	0.8				-0.43				N.A.			
		Biodiversity	-0.5	0.3	0.15	0.5				0.08				N.A.			
		Recreation	-0.1	0.4	0.04	0.6				0.02				N.A.			

	Timber	0.6	0.9	-0.54	0.8	-0.8	0.7	-0.43	0.43	-0.38	N.A.	S.Q.	N.A.
Government	Biodiversity	-0.5	1	0.5	0.8	-0.6	0.6	0.40	-0.30	0.30	Sub.	N.A.	Inf.
	Recreation	-0.1	0.6	0.06	0.8	-0.6	0.8	0.05	-0.04	0.05	N.A.	N.A.	N.A.